Universitat Autònoma de Barcelona

Improving governance for biodiversity conservation: ecosystem services and stakeholder participation in protected areas

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Summary

The dominant approach to biodiversity conservation stresses its connection with human well-being and sustainable development. To this end, the notion of ecosystem services is proposed. The adoption of an ecosystem services approach, together with a shift towards participatory approaches in conservation, has generated considerable debate. This thesis attempts to contribute to this debate by considering these approaches in decision-making on protected areas, and examine if they help to achieve effective biodiversity conservation and environmental justice. In addition, this dissertation attempts to advance knowledge about what type of policy instruments can best deal with biodiversity conservation and ecosystem services provision in protected areas. These various issues are elaborated in five essays, addressing the natural park of Sant Llorenç del Munt i l'Obac, located in Catalonia, Spain. The approach adopted in the thesis is multidisciplinary, involving a combination of qualitative and quantitative techniques.

The first essay proposes a framework for studying the relationship between biodiversity, ecosystem services and conservation policy. It argues that to address effectiveness well, the analysis of biodiversity policy needs to account for unwanted, avoidable indirect effects of biodiversity policy that cause the policy to be less effective than is feasible. A characterization of five types of these indirect effects – i.e. rebound – is provided. The second essay shows that the notion of ecosystem services is useful for integrating values of stakeholders into protected areas decision-making. This is examined by undertaking a sociocultural valuation of ecosystem services provided by the protected area. The results show that people are aware of the wide range of ecosystem services provided. Among these, habitat and cultural services are the most valued. Age, education and place of residence are the main characteristics of respondents that affect their sociocultural valuation. Conflicting viewpoints among stakeholders are deemed necessary to be included in decision-making. The third essay investigates the usefulness of social network analysis as a tool to support the creation of a broad representation of stakeholders in participatory processes. It assesses the structure of the social network of communication among stakeholders of the protected area, and compared this with the formal participatory bodies of the protected area. The fourth essay explores the implications of participatory arrangements in transforming power relationships in the governance of protected areas. It finds that the implementation of participatory arrangements has led to the exclusion of key social actors from the decision-making process associated with the protected area and has favoured the inclusion of actors motivated by economic interests. This process illustrates what has been called a "neoliberal approach" to biodiversity governance. The fifth and final essay evaluates a recently developed policy instrument aimed at promoting sustainable tourism by establishing a voluntary commitment between the managers of protected areas and relevant stakeholders. This involves comparison with instruments adopted in case studies located in four other European countries.

Chapter 1. Introduction

1.1 Motivation and aim

The loss of biodiversity worldwide is alarming (Butchart et al., 2010), with 12% of bird species, 23% of mammals, 32% of amphibians and 25% of conifers being currently threatened with extinction (MEA, 2005). Studies indicate that rates of biodiversity loss will continue, or accelerate, over the 21st century (Pereira et al., 2010). As an example, estimates of vertebrate extinctions range from 11 to 34% if a scenario of 0.8° to 1.7°C of global warming is considered (Thomas et al., 2004). The drivers of biodiversity loss are many, from direct causes such as hunting, habitat destruction, environmental pollution, climate change, invasive species, desiccation, to fundamental causes of an economic, demographic, technological and institutional nature. The latter take the form of, among others, inadequate institutions and governance systems (Rands et al., 2010).

One of the main strategies for ensuring biodiversity conservation is the establishment of protected areas which currently cover 12% of Earth's land surface (Butchart et al., 2010). The approved Strategic Plan for Biodiversity of the Convention on Biological Diversity (CBD) proposes to increase land under protection by 2020 globally to at least 17% of total terrestrial land area (CBD, 2011). However, studies indicate that the global protected-area network is far from fully achieving its main objective focused on preserving biodiversity (Rodrigues et al. 2004; Brooks et al., 2004). In the case of the European Union (EU), a Biodiversity Strategy has been adopted which focuses on halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020. The Natura 2000 network established in the EU is the largest network of protected areas, covering almost 18% of the EU territory (Sundseth, 2012). Nevertheless, biodiversity remains under acute pressure in the EU mainly due to human activities, with up to 25% of European animal species facing extinction and 65% of habitats of EU importance in unfavourable conservation status in 2010 (EEA, 2010) and continues getting worse (EC, 2015).

Nowadays, the dominant approach to conservation strengthens the link between biodiversity conservation, human well-being and sustainable development. It is widely adopted in international initiatives such as the Convention on Biological Diversity (CBD, 1992), the Millennium Development Goals (Sachs et al., 2009), the Millennium Ecosystem Assessment (MEA) (MEA, 2003) and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz et al., 2015). It recognises the necessity of community participation in conservation (Dudley, 2008; Borrini-Feyerabend et al., 2013). Ecosystems are increasingly regarded as complex adaptive systems of which humans are an integral part. Conservation need to be understood then as an interdisciplinary science focused on social ecological systems (Berkes et al., 2003; Díaz et al., 2015).

The notion of ecosystem services was proposed to capture this conceptualization of the links between biodiversity conservation and human well-being (Goldman et al., 2008. Ecosystem services are defined as the direct or indirect contributions of ecosystems to human benefits or satisfaction of needs (MEA, 2003). Concern is rising that loss of biodiversity may have severe consequences for ecosystem functioning and provision of services, as well as for associated aspects of human well-being (Balvanera et al., 2006; MEA, 2003; Cardinale et al., 2012; Cimon-Morin et al., 2013). However, the adoption of the ecosystem services approach in biodiversity conservation generates lively debates among scientists (Mace, 2014; Sandbrook, 2015) over how much emphasis ecosystem services should receive (Armsworth et al., 2007). This thesis analyses the implications of both adopting an ecosystem services approach and promoting stakeholder participation in governance of protected areas and in policy instruments implemented in terms of effective biodiversity conservation and the promotion of environmental justice – clarified in the next section.

One of the main challenges in adopting an ecosystem services approach to conservation is the complex relationship between biodiversity conservation, ecosystem services and policy. There is growing consensus among ecologists that, in general, biologically diverse ecosystems provide a greater flow of ecosystem services than non-diverse systems (Hooper et al., 2005; Lavorel et al., 2011; Flombaum and Sala, 2008; Balvanera et al., 2006). However, although biodiversity conservation policy is increasingly justified based on the ecosystem services provided, there is little empirical evidence that there exists a direct relationship between biodiversity conservation and ecosystem services delivery. In fact, the little quantitative evidence available to date has led to mixed conclusions (Chan et al., 2006; Naidoo et al., 2008). For instance, studies show that certain ecosystem services such as provisioning of clean water (Brisson & Chazarenc, 2009; Cardinale et al., 2012; Febria et al., 2015), provisioning food, fibre, fodder, and fuel (Cimon-Morin et al., 2013), regulating water flow, carbon sequestration (Leisher, 2015; Putz and Redford, 2009), and soil retention (Zheng et al., 2008) have a minimal dependence on biodiversity. Considering these results, some scientists raised concerns that focusing on ecosystem services might go at the expense of biodiversity conservation (McCauley, 2006; Putz and Redford, 2009; Vira and Adams, 2009; Adams, 2014; Leisher, 2015). Without a good knowledge of the relationship between biodiversity and provision of ecosystem services, we risk developing ineffective policies characterized by unwanted trade-offs and unexpected indirect effects on biodiversity and other environmental elements. This thesis explores the relationship between biodiversity and provision of ecosystem services and characterizes the potential ineffectiveness of conservation policies in terms of unintended, unwanted and avoidable indirect effects.

A particular challenge of the ecosystem services approach is how to value the contributions of ecosystems to human well-being. Three different valuation approaches of ecosystem services have been proposed, namely biophysical, economic and sociocultural. The dominance of monetary valuation in the ecosystem services literature (Vihervaara et al., 2010; Nieto-Romero et al., 2014) has led to debates about the implications of valuing nature in monetary terms (Kallis et al., 2013; Gsottbauer et al., 2015). These concerns include the reduction of the complexity of human behaviour to consumer preferences (Spash, 2008b; Spangenberg and Settle, 2010), potential erosion and crowding-out of noneconomic justifications for conservation (Redford and Adams, 2009; Doak et al., 2013; Neutreleers and Engelen, 2015), and the potential commodification of nature (Kosoy and Corbera, 2010; Gómez-Baggethun et al., 2010a; Gómez-Baggethun and Ruiz, 2011; Robertson 2012; Kelemen et al., 2014). The notion of value pluralism considers that more dimensions of nature's value, beyond economic value, are required in order to capture the multiple ways in which ecosystem services contribute to human well-being. These are social, cultural, ecological, spiritual, therapeutic, relational and place values (Kumar and Kumar, 2008; Gómez-Baggethun and Martín-López, 2015). It is against this background that sociocultural valuation has been promoted. Sociocultural valuation, also known as non-monetary approaches, examine the importance, preferences, needs or demands expressed by people towards nature, and articulate multiple values that people assign to nature through qualitative and quantitative measures others than money (Chan et al., 2012a, 2012b; Martín-López et al., 2014). Few concrete applications have been done (see Martín-López et al., 2012; Calvet-Mir et al., 2012; Oteros-Rozas et al., 2014). However, Gsottbauer et al. (2015) consider that non-monetary approaches have not offered very specific information to environmental policy making and do not allow for robust quantification. Thus, undertaking more sociocultural valuation studies is useful as it can add insight about such alternative valuation exercises. This research is based on the belief that the benefits protected areas provide to human well-being need to be explicitly accounted for in order to contribute to adequate conservation. We consider the concept of ecosystem service as a useful tool to shed light on such benefits. In this thesis sociocultural valuation is used to asses them. Identifying diverging preferences of stakeholders regarding ecosystem services is undertaken to allow designing effective and equitable nature management regimes.

Adequate institutions and governance structures that represent the aforementioned multiple values of nature (Gómez-Baggethun and Martín-López, 2015) are needed to ensure biodiversity conservation and the provision of ecosystem services (Muradian and Rival, 2012; Díaz et al., 2015). Several authors regard participation of stakeholders to include various forms of knowledge in governance of protected areas as an important element of this (Brown, 2003; Dudley, 2008; Bodin and Crona, 2009; Primmer and Furman, 2012; Borrini-Feyerabend et al., 2013). However, although appropriate governance and institutional structures determine conservation success of biodiversity policies, many policies generally are failing in this respect (Smith et al., 2009; Rands et al., 2010). One challenge is to ensure representativeness of different stakeholders in governance structures. The problem is that stakeholders are usually identified and selected through a subjective assessment that is sensitive to issues of relative power, influence and legitimacy, easily leading to a misrepresentation of stakeholders (Frooman, 1999). Moreover the role communication networks can play in categorizing and understanding stakeholder relationships is often overlooked (Prell et al., 2009). Thus, there is the need to develop methods that ensure an accurate evaluation of stakeholder representativeness in biodiversity governance. This thesis undertakes a social network analysis in order to disclose the actual communication network of stakeholders of the protected area, thus providing insight into the social structure of stakeholders.

Involvement of stakeholders in conservation decision-making has led to a restructuring of biodiversity governance in Europe, especially during the last two decades (Paloniemi et al., 2015). This has given rise to various network-based forms of governance. However, in practice, these new forms of governance do not always mean enhancing democracy, empowering citizens, or more effective governance (Swyngedouw 2005; Rauschmayer et al., 2009). As Harvey (2005) argues, this realignment from top-down state governing alone to networked governance by a broader configuration of the state-civil society-markets ensemble has been marked by neoliberalism. In the case of environmental governance it has involved a reconfiguration of the institutional arrangements charged with managing nature to include or sometimes favour market-based actors and practices (Bridge and Perreault 2009). In this context, ideas such as profitable public–private partnerships and increased business involvement are often promoted (Apostolopoulou et al., 2014), as is happening in the implementation of Natura 2000 network (Ferranti et al., 2014). These processes have been intensified recently, especially since the economic crises and the budgetary cuts, and have been analysed in Greece and UK (Apostolopoulou and Adams, 2015; Apostolopoulou and Pantis, 2010). There is therefore a need for more studies to examine the implications of these new participatory governance arrangements in terms of biodiversity conservation and environmental justice (Apostolopoulou et al., 2014; Brockington and Wilkie 2015), particularly in the context of an ongoing economic crisis (McCarthy, 2012). This thesis investigates the political dimension of stakeholder participation in protected area governance in the context of an increasing adoption of a neoliberal approach in biodiversity governance.

Due to the complexity and multi-layered character of the governance of ecosystems services and biodiversity conservation, there is a challenge to reinforce hybrid regimes based on governmental command-and-control, economic instruments and community-based institutional arrangements (Ostrom, 2005; Rands et al., 2010; Muradian and Rival, 2012). This goes together with the policy mix concept (Ring and Schröter-Schlaack, 2011) promoting a combination of policy instruments for biodiversity conservation and ecosystem service provision. This thesis evaluates a policy instrument designed to promote sustainable tourism in protected areas as part of a broader policy mix on the basis of relevant criteria. It is compared with other instruments dealing with biodiversity conservation and the provision of ecosystem services implemented in Europe. This evaluation provides practical guidelines and draw lessons for better design of biodiversity policy.

The thesis focuses on the natural park of Sant Llorenç del Munt i l'Obac, located in the northeast of Spain, within the Catalan precoastal mountains. The natural park represents an appropriate study case because its management integrates biodiversity conservation, public use and the promotion of economic activities among inhabitants of villages in and around the protected area. It further has a long tradition of governance with participation of local stakeholders. The results and conclusions of this thesis will help to improve governance structure and the design of biodiversity policy in protected areas by careful balancing of biodiversity conservation, provision of ecosystem services and environmental justice.

1.2 Conceptual-theoretical framework

Three main theoretical frameworks have guided this thesis, namely the ecosystem services approach to conservation, institutions and governance structures in conservation, and political ecology of conservation. In this section I briefly present them and provide information that is not already in the different chapters of the thesis.

1.2.1 Ecosystem services approach in conservation

In recent decades, humanity's reliance on the natural environment has increasingly been expressed through the concept of ecosystem services (Redford and Adams, 2009). Ecosystem services are defined as the direct or indirect contributions of ecosystems to human benefits or satisfaction of needs (TEEB, 2010b). Examples of these services are food provision, clean air or ecological knowledge. The reasoning is that biodiversity influences ecosystem functioning in a complex way, which determines the provisioning of services that affect human well-being (MEA, 2005). Furthermore, human well-being is a complex issue which has multiple constituencies, including basic material for a good life, freedom and choice, health, good social relations, and security. The constituents of well-being, as experienced and perceived by people, are situation-dependent, reflecting local geography, culture and ecological circumstances (MEA, 2003). This concept of ecosystem services took wings in the late 1990 with publications such as Costanza et al. (1997) in *Nature*, Vitousek et al. (1997) in *Science* and Daily (1997) book *Nature's Services: Societal dependence on Natural Ecosystems.* Few years later, the MEA (2005) and the TEEB (2010b) have mainstreamed this concept in conservation and environmental policy. The proponents of the concept of ecosystem services consider that it can be a powerful incentive for conserving nature through showing its instrumental value and its relevance for human well-being (Balmford et al., 2002; Goldman et al., 2008; Mace et al., 2012). Some scientists consider necessary to value ecosystem services in monetary terms to pursuit win-win solutions both for biodiversity conservation and firms because it allows integrating the value of nature's benefits into their operations in order to ensure sustainable development (Daily, 1997; TEEB, 2010b; Bateman et al., 2011; Marvier and Kareiva, 2014). However, concerns about the implications of expressing the value of ecosystems in monetary terms have reached claims on the necessity to strengthen other approaches. Actually, three different valuation approaches have been proposed, namely biophysical, economic and sociocultural. The first approach assesses value based on intrinsic properties of objects by measuring underlying physical parameters, whereas the other two approaches assess value based on preference-based approaches (Pascual et al., 2010).

1.2.1.1 Sociocultural valuation of ecosystem services

Sociocultural valuation is proposed as an approach to examine the importance, preferences, needs or demands expressed by people towards nature for their well-being (Chan et al., 2012a, 2012b; Martín-López et al., 2014). It is based on expressing lexicographic preferences and establishing rankings of ecosystem services instead of expressing importance through another entity, being money, energy or labour, such as in monist value theories (Spangenberg and Settele, 2010). Sociocultural valuation is aimed to capture the multi-dimensional nature of value when referring to ecosystems (Martín-López et al., 2014; Kumar and Kumar, 2008), including less tangible social and ethical concerns of ecosystems usually associated with non-material benefits such as the satisfaction of conserving biodiversity, local identity, or local ecological knowledge (Chan et al., 2012a, 2012b). In this sense, moral, ethical, historical or social aspects play an important role (Castro et al., 2014), which frequently have an incommensurable character (Martínez-Alier et al., 1998; Chan et al., 2012b; Martín-López et al., 2014). Relational values are increasingly deemed important and need to be added to the already acknowledged instrumental values – protecting nature for humans' sake – and intrinsic[1] values – protecting nature for nature's sake (Chan et al., 2016). Relational values are preferences, principles, and virtues associated with relationships, both interpersonal and as articulated by policies and social norms. It entails reflection on the appropriateness of preferences, emphasizing that value is derived from a thing's or act's contribution to a good life. People therefore make choices considering core values such as justice, care, virtue, and reciprocity and therefore considering the appropriateness of how they relate with nature and with others, including the actions and habits conducive to a good life. Socio-cultural values therefore are not limited to cultural ecosystem services alone and should be connected to the full spectrum of them which includes provisioning,

[1] Intrinsic values are another value concept, without a subject, so non-transformable into monetary units, but instead taking the form of a right (Nunes et al., 2003).

regulating, habitat and cultural ecosystem services (Scholte et al., 2015). Moreover, sociocultural valuation can reveal the importance people attribute to ecological systems as a whole. Whereas economic valuation of ecosystem services is based on the intensity of changes in people's preferences under small or marginal changes in the quantity or quality of goods or services (Pascual et al., 2010; Nunes et al., 2003), in sociocultural valuation people can reveal the attached importance to the system – or ecosystem services – as a whole.

Although some classifications of sociocultural valuation methods have been proposed (Castro et al., 2014; Scholte et al., 2015; Kelemen et al., 2014), a consistent classification is still lacking due to the recent emergence of this literature. There is the need, therefore, to advance in studies dealing with sociocultural valuation to develop a consistent classification of methods and clarify the foundations of this new approach (Gómez-Baggethun et al., 2014).

1.2.2 Institutions and governance structures in conservation

Institutions and governance structures are considered indirect drivers of change in ecosystems because usually they do not affect nature directly, but rather through their effects on direct anthropogenic drivers (MEA, 2003; Díaz et al., 2015). Institutions encompass all formal and informal interactions among stakeholders and social structures that determine how decisions are taken and implemented, how power is exercised, and how responsibilities are distributed (Ostrom, 1990; 2005). Examples of institutions are systems of property and access rights to land, legislative arrangements, treaties, customary laws, informal social norms and rules, and international regimes. Institutions play a significant role in influencing people's perception about the importance of nature's benefits and their behaviour and thus decisions about the way they interact with nature.

The current understanding of conservation[2] shows that it has evolved from the traditional framing considering that nature should be protected from human action, known as fortress conservation, to frames beginning in the 1970 which considered the need of humans' inclusion in conservation, namely community-based conservation/people-centred conservation (Adams and Hulme, 2001; Berkes, 2004;

[2] A dominant understanding of conservation is reflected by the so-called Global Strategy (WRI, 1992): "the management of human use of the biosphere so that it may yield the greatest sustainable benefit to current generations while maintaining its potential to meet the needs and aspirations of future generations: thus conservation is positive, embracing preservation, maintenance, sustainable utilisation, restoration, and enhancement of the natural environment". The Global Strategy is composed by the World Resources Institute (WRI), the International Union for the Conservation of Nature (IUCN), the United Nations Environmental Programme (UNEP), the Food and Agriculture Organization of the United Nations (FAO) and UNESCO.

2007; Brown, 2003). Community-based conservation considered that conservation must be participatory including the viewpoints of people living in protected areas and attributed increasingly importance to social and economic factors as determinants for conservation success (Ghimire and Pimbert 1997). Hence, stakeholders' participation and the inclusion of various forms of knowledge in governance of protected areas has been deemed important for biodiversity conservation (Brown, 2003; Dudley, 2008; Bodin and Crona, 2009; Primmer and Furman, 2012; Borrini-Feyerabend et al., 2013). Using knowledge and perspectives from the community level can help build a more complete information base than the one available only from scientific studies (Berkes et al., 2000). In fact, excluding local people from conservation strategies not only threatens their livelihoods but also the biodiversity that is to be conserved, since what are thought to be pristine ecosystems are actually cultural landscapes that depend on particular management practices (Reyes-García et al., 2013). It further disrupts long processes of storing and transmission of local ecological knowledge systems that have great value for sustainable ecosystem management and biodiversity conservation, not only in developing countries but also in industrialized ones (Gómez-Baggethun et al., 2010b, 2010c; Otero et al. 2013).

The establishment of relationships of communication among stakeholders is very important for good participatory governance because they guarantee access of information, exchange of knowledge and the building of relations of trust (Borgatti and Foster, 2003; Rishi, 2007; Bodin and Crona, 2009). These connections among stakeholders constitute social networks, which are observable social structures made up of individuals or organizations tied by one or more specific types of interdependency, such as common interests or communication exchange (Bodin et al., 2006).

1.2.3 Political ecology of conservation

Political ecology is a field focused on investigating human-environment relations, which are considered deeply and inextricably linked (Robbins, 2004). These relations are shaped by economic, social and political power determining distributional issues (Roth, 2015). It emphasizes that the actual state of nature needs to be understood materially as the outcome of political processes, which in turn affect the way nature itself is understood (Adams and Hutton, 2007). Political ecology examines the intersection of socio-economic context, political relations, cultural practices, and ecological processes to highlight how particular environmental governance and management regimes become dominant and how this affects nature-society relations (Bryant, 2001; Walker, 2003; Neumann, 2005). Moreover, political ecology is an explicitly normative approach committed to environmental justice and sustainable future, seeking not just to analyse struggles over resources but also to influence them (Forsyth, 2008; Roth, 2015). It is a diverse and transdisciplinary field which has changed through time in response to dominant issues and theoretical paradigm shifts (Bryant, 2001).

Several aspects characterize the political ecology of conservation. First, it involves an understanding of nature as necessarily socially produced (Castree and Braun, 2001; Roth, 2015). This means that it is shaped by human action, which is influenced by human ideas about what 'nature' is and how 'nature' should be. In this sense, protected areas reflect how environment-humans relation is understood (West et al., 2006; Adams and Hutton, 2007). Political ecologists have studied the history of protected areas and its implications for local communities and have criticized forms of conservation aimed to exclude stakeholders from protected areas and instead supported those initiatives that treat landscapes in their social, ecological, economic and cultural dimensions (Roth, 2015).

The second feature is an explicit focus on how processes of power shape conservation practice and thus its social and ecological outcomes (Roth, 2015). This thesis specifically draws on the literature on political implications of biodiversity governance, particularly the role of emerging governance forms based on participatory arrangements. Participation involves sharing of power between the governed and the government, and thus involves various actors and power relations, structural inequalities and different class, ethnic, cultural and gender groups. Political ecology investigates who benefits and who does not benefit from conservation initiatives and considers the multiple interests and actors within communities, how these actors influence decision-making over time, and the internal and external institutions that shape the decision-making process (Agrawal and Gibson, 1999). There is a growing literature explicitly linking the emergence of this new governance forms with neoliberalism (Harvey, 2005) and particularly with a neoliberal shift in conservation (Igoe and Brockington, 2007; Büscher et al., 2012; Apostolopoulou et al., 2014). It consists of reconfiguring conservation institutional arrangements to favour market-based actors and practices leading to the establishment of profitable public–private partnerships and increased business involvement. Neoliberal conservation is an amalgamation of ideology and techniques informed by the premise that natures can only be "saved" through creating markets, private investments and its succeeding revaluation in economic terms (Büscher et al., 2012).

The third feature of political ecology of conservation considers that local-scale social dynamics need to be contextualized within a broader national, regional and global setting (Robbins, 2002; Walker, 2003; Neumann, 2009; Roth, 2015). In this sense, the governance of nature is evermore subject to interactions of different actors at varying scales, that is, the local is subject to different types of processes occurring at regional, national, and international levels through forces ranging from legislation, government programs, international treaties, and the market, among others. The recent, and ongoing, economic crisis brings new questions and urgency to these debates (McCarthy, 2012; Apostolopoulou and Adams, 2015).

1.3 Case study: the natural park of Sant Llorenç del Munt

The natural park of Sant Llorenç del Munt (hereafter Sant Llorenç) is located in the north-east of Spain, within the Catalan pre-coastal mountains (Figure 1.1). It corresponds to category V, i.e. “Protected Landscape”, of the IUCN classification of protected areas. This category is created to protect the ecological, biological, cultural and scenic value of these areas (IUCN, 1994). It covers 13694 hectares, and comprises 12 municipalities and is managed by the Diputació de Barcelona, a regional administration corresponding to the territorial area of the Barcelona Province. It consists of a mixture of private (59,20%) and public land ownership – mostly public land owned by Diputació de Barcelona and the remainder by the Generalitat de Catalunya (Catalan government) (Diputació de Barcelona, 2012).

The geology of the zone consists mainly of siliceous and carbonate conglomerates intercalated with sandstones and lutites of continental origin deposited during the Eocene. The climate is mild Mediterranean, and the annual rainfall ranges between 675 mm in the town centre of Matadepera (423 m a.s.l.) and 850 mm in the mountain peak (1107 m a.s.l.) (Martín and Moreno, 1994). The vegetation is the typical Mediterranean one, composed by sclerophyllous forests and scrublands with typical communities such as oaks, shrubs and weeds. Vegetation is composed of holm oak woods (*Quercus illex),* mixed with Aleppo and Stone pines (*Pinus halepensis* and *P. pinea*) in the lower parts of the mountain. There are also communities of rupiculous plants on lithological outcrops and some deciduous trees such as hazelnut (*Corylus avellana*) in the northern and most humid slopes (Nadal et al., 2009). Settlement has historically been structured in dispersed *masos* (homesteads) that combined subsistence agriculture based on cereals, vines and olive trees with the exploitation of forests. By the end of 19th century wealthy people from nearby cities started to spend their summertime in *masos* or in surrounding villages (like Matadepera), which gained importance as a summer place for industrial burgeois (Otero, 2010).

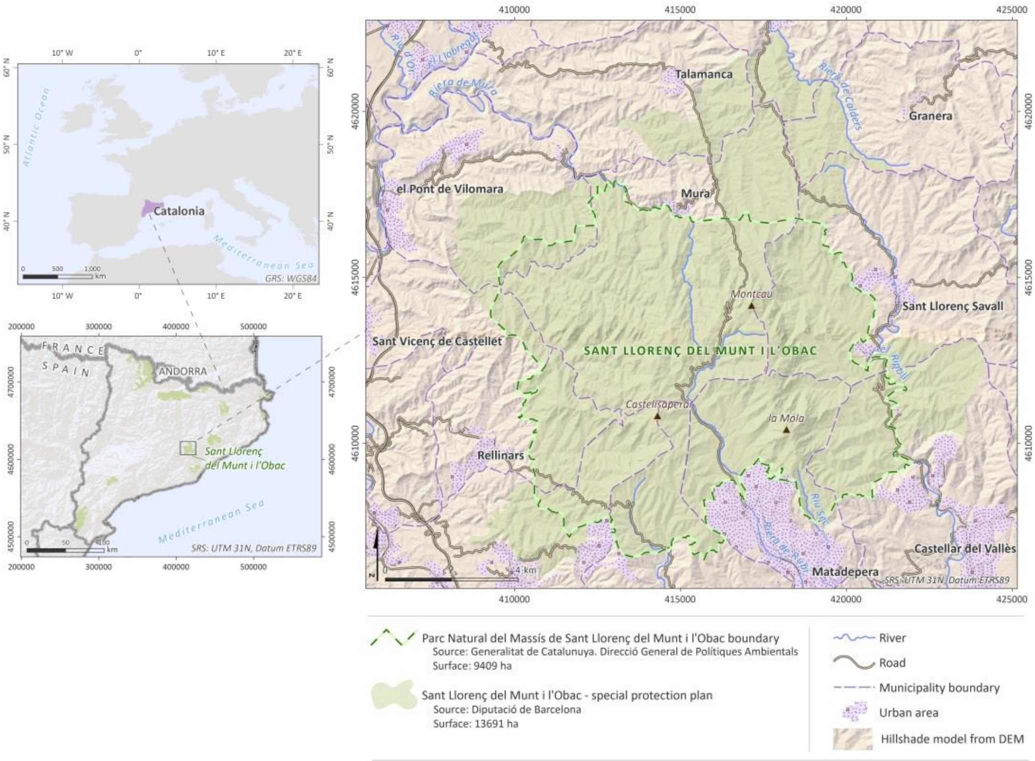

Figure 1.1 Map of the study area

The park is surrounded by large cities from the Barcelona Metropolitan Region, which implies a high frequency of visitors. Only in weekends in 2011 more than 120000 people visited the park, of which 79% were concentrated in its central area (Diputació de Barcelona, 2012). Traditional activities have nearly disappeared (e.g., exploitation of oak to produce charcoal) or have greatly diminished (e.g., agriculture, logging and cattle grazing). Currently the service sector is the largest activity in the park, mainly for tourism. During the last decades, the main trends occurring in socio-ecological terms are growth of the urban area in Matadepera, abandonment of traditional rural activities, growth of forest cover and an increase in the risk of fires (Otero, 2010).

A brief history of protection measures in the park area

The natural park of Sant Llorenç was initially protected in 1972 by the Pla Especial d'Ordenació (Special Plan). The protected area was composed of two sub-areas: one above 800 meters a.s.l. where urbanization was forbidden (2655ha in size); and a zone of influence below 800 meters a.s.l. where urbanization was permitted (4500ha). The initial protection of the area under Franco's dictatorship (1939-1975) reflected more a strategy of legitimizing residential areas than a real interest to protect nature. The establishment of protected areas was really part of a broader strategy of the dictatorship to improve its international damaged reputation (Aguilar, 2012). The high social and ecological value of the area and the threat of urban spread were the reasons that promoted an intense demand for protection by a conservationist movement arising at the end of the seventies. This movement was mainly structured around hiking groups present in most villages and cities in the surroundings of the protected area and was highly influenced by the political and social context of the democratic transition period after the dictatorship (Aguilar, 2012). After protests, in 1982 the new democratic local governments revised the plan of 1972 and extended the protected area in which urbanization was forbidden to 9638 ha. It was designed officially as a Natural Park by the Catalan government in 1987 (Estany et al., 2010). Villagers initially complained about the restrictions of the protected area, such as bans on lighting fires, harvesting thyme or hunting birds. However, during the following years they accepted the measures as they understood it was beneficial to conservation. Meanwhile, the conservationist movement expressed the opinion that good conservation of an area implied entirely forbidding human activities in it. However, over time came to the understanding that cultural landscapes need traditional human activities for their maintenance and sustainable management (Aguilar, 2012).

1.4 Research objectives and questions

This thesis aims to analyse the implications of the ecosystem services approach and participation of stakeholders in protected area management in order to achieve sustainable and environmentally just management of natural resources, notably biodiversity conservation. The research is guided by four research objectives. The first one is to identify the unwanted indirect effects of biodiversity policy originated due to the complex relationship between biodiversity conservation, ecosystem functioning and ecosystem services. The second objective is to show how the ecosystem services approach can contribute to improving protected area management. We hypothesize that understanding better social perceptions and values that people attribute to protected areas – including conflicting ones between individuals – can contribute to better management. The third objective is to evaluate stakeholder representativeness in the governance of protected areas. The fourth objective is to analyse the political implications of participatory arrangements in protected area governance. The fifth objective is to advance knowledge on the appropriate type of policy instrument to assure biodiversity conservation, ecosystem services provision and integration of diversity of stakeholders.

The following main research questions will be addressed:

1) Which types of unintended, unwanted and avoidable indirect effects of biodiversity policy can cause such policies to be less effective in terms of conservation benefits?

2) Which are the ecosystem services provided by the natural park of Sant Llorenç and what is the importance attributed to them by visitors and inhabitants? Which are the factors that influence individuals' valuation of ecosystem services?

3) What is the structure of the social network of communication between stakeholders associated with the natural park? Is this network of communication between stakeholders represented in the formal participatory bodies of the park management?

4) Why was stakeholder participation promoted in the management of the natural park of Sant Llorenç? Which topics are being addressed in the participatory process and in which manner? Who are included and excluded in/from the participatory process?

5) Which type of policy instruments can best assure biodiversity conservation, ecosystem services provision and integration of diversity of stakeholders?

1.5 Outline of the Thesis

The thesis is structured in five chapters to elaborate the research questions mentioned above. Chapter 2 explores the relationship between biodiversity, ecosystem services and conservation policy. For this purpose it develops a framework for studying their interdependence. It is argued that a necessary (though not sufficient) condition for making a transition to a truly sustainable economy is that biodiversity conservation and its analysis take into account unwanted and avoidable indirect effects of all kinds of biodiversity policy. These indirect effects partly undo the direct conservation benefits causing the policy to be less effective than is possible. Five types of these indirect effects are identified – which are called rebound effects– and the terms biodiversity (two types), ecological, service and environmental rebound are proposed for these. The service rebound is associated with the problem of incongruence or conflicts, and thus the potential need for trade-offs, between ecosystem services or between such services and biodiversity conservation. Effective biodiversity policy requires the minimization of these various rebound effects.

Chapter 3 investigates the usefulness of the notion of ecosystem services for integrating perceptions and values into decision-making on protected areas. Socio-cultural factors have been suggested to explain perceptions and values assigned to ecosystem services. The chapter examines this by undertaking a socio-cultural valuation of ecosystem services provided by the natural park of Sant Llorenç. Four methods are used, namely a review of the literature on ecosystem services, non-participant observation, semi-structured interviews with stakeholders, and a valuation survey among visitors. The chapter assesses whether visitors and other stakeholders understand the term ecosystem service, finding that the concept is rather unknown or misunderstood. Among the 28 ecosystem services identified, habitat and cultural services were the most valued. The analysis involves statistically identifying socio-economic characteristics of visitors that have a main influence on their valuation of ecosystem services. It further assesses diverging preferences of all stakeholders that might give rise to conflicting views about policies for protected areas. Lessons are drawn about the usefulness of the multi-method approach and about the management of protected areas.

Chapter 4 is motivated by the idea that local participation of stakeholders in governance of protected areas is important to natural resource management and biodiversity conservation. For this purpose, social network analysis (SNA) is proposed as a useful tool for analysis because it allows the understanding of stakeholders' relationships, interactions, and influences through communication networks. This chapter combines quantitative and qualitative data to undertake a SNA for the natural park of Sant Llorenç. This is aimed at assessing the structure of the communication network; comparing the informal communication network with the formal participatory bodies of the natural park; and selecting participants for subsequent analyses of the adequate governance structure of the natural park. The results suggest that an informal network of communication, which is reasonably well represented in participatory bodies, exists. However, this communication network is not functioning perfectly because stakeholders experience a lack of trust in the governance bodies of the park, which they perceive to be ineffective. The results show that SNA is an effective tool to support the creation of a broad representation of stakeholders in participatory processes.

Chapter 5 investigates the political implications of participatory arrangements in the governance of protected areas. In many countries, network-based forms of governance have appeared in the process of restructuring biodiversity governance over the last two decades. This has promoted the involvement of relevant stakeholders. Even though this shift has been framed as promoting stakeholder and public participation, in practice it has often hampered democratic decision-making and community empowerment by reinforcing business involvement, particularly through the creation of profitable public–private partnerships. This chapter examines these processes by analysing the role of participatory arrangements in the governance of the natural park of Sant Llorenç. In particular, it studies how these have transformed power relationships. For this purpose, it assesses the reasons for promoting stakeholder participation, what issues are addressed in the various participatory bodies, and the specific procedures and inclusiveness characterizing these bodies. The chapter finds that participatory arrangements paradoxically led to exclusion of certain key social actors in the management of protected areas. They further facilitated a neoliberal approach to biodiversity governance by favouring the inclusion of actors with mainly economic motivations.

Chapter 6 assesses the European Charter for Sustainable Tourism (ECST), a policy instrument implemented in the natural park of Sant Llorenç. The ECST is aimed at promoting sustainable tourism by establishing a voluntary commitment between managers of protected areas and relevant stakeholders. We assess this policy considering as performance criteria effectiveness, efficiency, equity and legitimacy, among others. As part of effectiveness, various types of unwanted and avoidable indirect, rebound effects are considered. The ECST is analysed as part of a policy mix, that is, a combination of different types of policy instruments. This chapter further compares the performance of the ECST with policy instruments used in four other European countries and identifies rebound effects of such instruments. Considering the results of comparing instruments, several recommendations are made that can improve biodiversity conservation policies.

Finally, chapter 7 draws specific and general conclusions, and suggests some avenues for further research.

Chapter 2. Ineffective biodiversity policy due to five rebound effects[3]

2.1 Introduction

Much has been written on biodiversity policy, from the perspective of biology, ecology, economics and policy sciences. What is missing in most writings is serious attention for the potential ineffectiveness of such policies in terms of unintended, unwanted and avoidable indirect effects. Effectiveness of biodiversity policy can be interpreted in various ways, namely in terms of biodiversity conserved, ecosystem functions (functional diversity) maintained, or ecosystem services guaranteed. Effective biodiversity policy is a necessary condition for making a transition to a truly sustainable economy. In order to develop our thinking about this issue we propose a framework that connects various types of diversity, ecosystem functions and services, values and biodiversity protection policies. This framework will allow us to identify potential unwanted, avoidable effects of biodiversity policies on each of these components. We refer to such effects here as rebound, inspired by the literature on energy conservation and rebound (Sorrell, 2007; van den Bergh, 2011).

The drivers of biodiversity loss include many, such as hunting, land use, deforestation, fragmentation due to infrastructure, water use causing desiccation, and environmental pollution with climate change as a very important case. In addition, loss is enhanced by existing policies in sectors like agriculture, infrastructure and fisheries. An example is subsidies for biofuel production that promote conversion of tropical forest to tilled fields, which may reduce the area with habitats that support unique biodiversity (Kinzig et al., 2011). The complex set of drivers of biodiversity loss makes the analysis of effective policy not easy.[4]

For addressing effectiveness well, the analysis of biodiversity policy needs to consider indirect, avoidable effects of biodiversity policy. We identify five categories of such effects, and propose the terms biodiversity (two types), ecological, environmental and (ecosystem) service rebound for these. These terms reflect that certain strategies aiming at conserving specific biodiversity have unintended effects which partly undo the direct conservation benefits, causing them to be less effective than is possible. These

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[3]This chapter also appears as: Maestre-Andrés, S., Calvet-Mir, L., van den Bergh, J.C.J.M., Ring, I., and Verburg, P. 2012. Ineffective Biodiversity policy due to five rebound effects. *Ecosystem Services* 1:101-110.

[4] Of course, to achieve a sustainable economy it would also be necessary to address other policies that negatively affect biodiversity, for instance in areas like energy, agriculture, fisheries and infrastructure.

rebounds can be reduced by appropriate design of policies and strategies. It is evident that effective biodiversity policy requires the minimization of these rebound effects.

Solving the rebound of biodiversity policies is not easy, however, as the ineffectiveness is not always transparent. Recognizing rebound will evidently depend on the precise interpretation of biodiversity and the type of biodiversity indicator used. Moreover, biodiversity-related rebound can follow different mechanisms, as exemplified by the five types of rebound of biodiversity policy. In order to understand the chain of cause-effect relationships from biodiversity through ecosystem function to ecosystem services, values and policy, and the place of the five types of rebound, we present the scheme in Figure 2.1. Its elements will become clear in due course. Note that the aggregate classification of ecosystem functions and services is identical because the latter are appropriated functions. This does not deny that at a more disaggregate level there can be a distinction between the two, i.e. some functions are not appropriated or do not directly generate a service to humans.

In line with the aim of an opening issue of a new journal, we want to raise relevant research questions – both in terms of research and policy – about this theme. This includes discussing different notions of biodiversity, their connection with ecosystem services, how to compare policy options, and the role of ecosystem valuation concepts and methods to assess biodiversity loss or protection. Our discussion aims at providing arguments for broadening the analysis of biodiversity policy design by considering various types of indirect or rebound effects. Ultimately, this may give rise to distinct and new views on effective policy options and instruments.

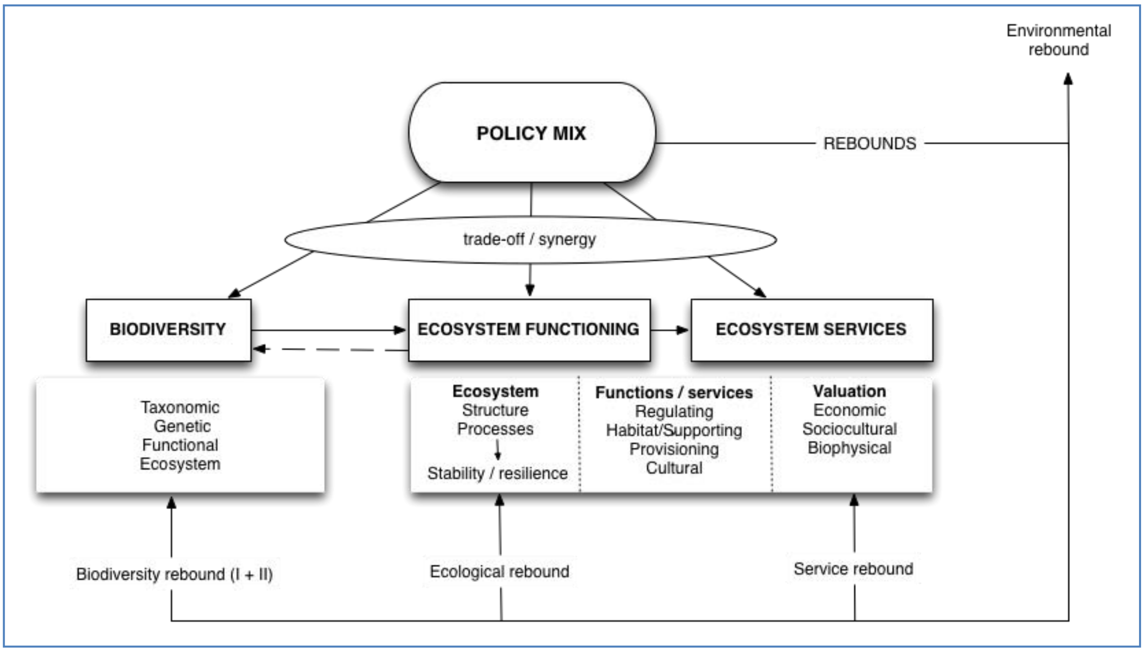

Figure 2.1 A framework for biodiversity policy, ecosystem services and rebound effects

2.2 Interpretations of biodiversity, ecological significance and policy relevance

Biodiversity has been defined as: ". . . the variability among living organisms from all sources, including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems" (Convention on Biological Diversity, 1992). So four fundamental facets of biodiversity can be distinguished, namely taxonomic, genetic, functional and ecosystem diversity. Taxonomic diversity refers to different taxa (e.g., class, order, family, genus, species). One specific type is species richness, that is, the number of species at a particular site or at a global scale. Genetic (or phylogenetic[5]) diversity is the genetic variation within and between species which is the fundamental level of diversity underpinning the other types. Functional diversity measures the number, type, and distribution of functions performed by organisms within an ecosystem, and thus reflects the diversity of morphological, physiological and ecological traits within biological communities and their interactions. It further indicates a degree of complementarity and redundancy[6] of co-occurring species (Díaz and Cabido, 2001; Hooper et al., 2005). Finally ecosystem diversity refers to the diversity of assemblages and their environments over a defined landscape, ecological zone or at global scale (Swift et al., 2004).

Another way of classifying biodiversity is based on spatial characteristics. A common distinction is based on the spatial focus of analysis being local or a habitat (alpha diversity) versus regional or a landscape (gamma diversity). In addition, a more contentious notion, beta diversity or spatial turnover, captures among-site components or number of sub-units (habitats) (Hooper et al., 2005). Sometimes the relation between these diversity notions is summarized as gamma = f (alpha, beta), but this involves implicit assumptions.

The design of a valuation context requires the choice of a spatial frame of analysis (Norton and Ulanowicz, 1992). Whereas biodiversity loss is usually discussed at a global or worldwide level, biodiversity valuation studies frequently address policy changes or scenarios defined at local, regional or national levels. Although this seems contradicting, it can be argued that biodiversity and its loss are relevant at multiple

[5] Phylogenetic diversity relates to the evolutionary history of a species (Faith, 1992).

[6] Species that are redundant for one ecosystem process may not be redundant for others and species considered redundant under certain environmental conditions may become important under changed conditions (Harrington et al., 2010).

spatial levels, from local to global, and that local loss of non-unique species sometimes implies a loss in biodiversity value (Hammond et al., 1995).[7]

There is debate about which dimension of biodiversity is better in order to analyse the diversity of an area. Traditionally, taxonomic diversity has been the more used indicator. However, it is now well recognized that functional and phylogenetic relationships are also important indicators of biodiversity (Strecker et al., 2011). Consequently, functional diversity is now often assumed to be a better predictor of ecosystem functioning than other measures of diversity (Hooper et al., 2005). Nevertheless, one should not forget that interactions and functional roles of species involve complex and often unknown aspects, suggesting that phylogenetic diversity might capture species assemblages better than functional diversity to explain ecosystem productivity (Devictor et al., 2010).

Recent studies demonstrate that regions of high taxonomic diversity may be incongruent with regions of high functional or phylogenetic diversity (Forest et al. 2007; Strecker et al., 2011). Moreover, Devictor et al. (2010) found that phylogenetic and functional diversities were uncorrelated in many cases. However, this depends on the spatial scale of study of biodiversity, i.e. alpha, gamma or beta diversity. For instance, using beta-diversity patterns (among sites), functional and phylogenetic diversity were found to be positively related to taxonomic diversity, while the correlation between functional and phylogenetic beta-diversity was even higher.

These patterns of (non-)congruence of distinct indicators suggest that species occurring locally may derive from regional species pools with similar as well as different biogeographical and evolutionary histories (Cumming and Child, 2009). Moreover, for a given regional pool, species may respond to environmental gradients in different ways, which affects the spatial distribution of functional and phylogenetic diversity and can generate a spatial mismatch between taxonomic, functional and phylogenetic diversities (Prinzing et al. 2008). However, strong environmental filters could restrict species composition to a relatively restricted range of functional characteristics, thereby limiting the degree of functional diversity capable of influencing different ecosystem properties (Grime, 2001). Increasing species richness would then just lead to a finer division of the available niche space rather than to greater functional diversity (Díaz and Cabido, 2001). Mapping beta-diversities reveals coherent transitional zones between regions with different pools of species, functional or phylogenetic diversity. This approach can thus help to identify and delimit ecological

[7] Stirling (2007) proposes to regard diversity as a multidimensional concept, consisting of three dimensions, namely variety, balance and disparity. Variety denotes the number of different identities (functions, physical appearances, or behaviours) in a population of elements. Balance (or equality) captures the distribution or frequency of the various identities. Disparity refers to the degree of distance or qualitative difference between the identities in a population.

boundaries around areas of particular interest. On its own, beta-diversity is however silent on the amount of diversity of a given region. For instance, high beta-diversity can be found in highly fragmented landscapes with low gamma diversities if few species or little functional or phylogenetic diversity is found in these landscapes. Therefore, gamma and beta diversities offer complementary information on biodiversity patterns (Devictor et al., 2010).

This is all not just interesting for theoretical reasons but also shows a clear connection with policy. For example, in protected areas networks areas having the highest taxonomic diversity were protected whereas areas having the highest phylogenetic and functional diversity received less protection. A similar analysis for beta-diversity revealed a different pattern: Areas having the highest beta-taxonomic and phylogenetic diversity values were well protected, while areas with the highest functional diversity received less protection (Devictor et al., 2010). Measuring each of these complementary biodiversity components is necessary for understanding ecosystem functioning in terms of the complete structure, composition and dynamics of natural communities. Associating these with the ecosystem services provided in the relevant area allows one to develop a systematic conservation planning that accounts for multiple aspects of biological diversity, reflecting taxonomic, functional, and evolutionary perspectives (Bello et al., 2010; Hooper et al., 2005; Strecker et al., 2011).

The mutual relation between biodiversity and ecosystems functioning

Biodiversity both responds to, and influences, ecosystem functioning (Holling et al., 1995; Hooper et al., 2005). Variations in ecosystem functioning can result from fluctuations in the environment from year to year, directional changes in conditions, abiotic disturbance, or biotic disturbance. Although there is no a priori reason to expect that different ecosystem properties have a single pattern of response to changes in different components of biodiversity, some studies show that the most important dimension of biodiversity which influences ecosystem functioning are species' functional characteristics. These include effects of dominant species, keystone species, ecological engineers, and interactions among species (e.g., competition, facilitation, mutualism, disease, and predation). In addition, comparisons of distinct ecosystems suggest that abiotic conditions, disturbance regime, and functional traits of dominant species have a larger effect on many ecosystem properties than species richness.

Hooper et al. (2005) summarized the main responses of ecosystem functioning to changes in species or functional diversity. As shown in Box 2.1, the patterns depend on "… the degree of dominance of the species lost or gained, the strength of their interactions with other species, the order in which species are lost, the functional traits of both the species lost and those remaining, and the relative amount of biotic and abiotic control over process rates …" (p. 9).

Box 2.1 Possible responses of ecosystem functioning to biodiversity change

1) Diversity might have no effect: changing relative abundance or species richness might not change process rates or pool sizes. Lack of response could occur for several reasons, such as primary control by abiotic factors, dominance of ecosystem effects by a single species that was not removed, or strong overlap of resource use by different species.

2) An increase in diversity originates a change in ecosystem functioning, associated with two main mechanisms: increasing species richness increases the likelihood that those key species, which have a dominant effect on ecosystem properties, are present; species or functional richness contributes to ecosystem properties through positive interactions among species.

3) An increase in diversity implies a saturating response in ecosystem functioning. This is the most commonly hypothesized pattern, where complementarity, facilitation, and sampling effects for high productivity (or other properties) are all expected to show a similar saturating average response as diversity increases.

4) Complementarity and selection or sampling effects are not necessarily mutually exclusive. There can be a continuum of diversity effects, ranging from the probability of sampling one dominant species to the probability of selecting several complementary species.

5) Ecologists disagree over whether sampling effects are relevant to natural ecosystems. Some ecologists argue that sampling effects are artefacts of certain experimental designs because of their dependence upon the debatable assumption that communities are random assemblages of species from the total species pool. Others assert that they are simply an alternative mechanism by which species richness might affect ecosystem properties in natural communities, pointing out that there are many stochastic factors that can influence community composition.

6) Adding trophic levels is expected to lead to more complex responses of ecosystem properties to a change in biodiversity.

Source: based on Hooper et al. (2005).

2.3 Linking ecosystem functioning, ecosystem services and biodiversity

There is a broad consensus in the scientific community about the relationship between ecosystem functioning and ecosystem services. Ecosystem functions can be defined as all aspects of the structure and processes of ecosystems with the capacity to produce services that satisfy human needs directly or indirectly (Hooper et al., 2005). Gómez-Baggethun and de Groot (2010) distinguish between potential benefits associated with ecosystem functions and real benefits, which are the potential ones concretized once they are used or enjoyed by people. One can regard ecosystem services somehow as a simplifying translation of ecological complexity to a limited number of functions and ecosystem services. Various classification of these have been proposed in the past (e.g., Turner et al., 2000), which have converged to a quite uniform view of main categories and detailed services. Influential classifications have been proposed by de Groot et al. (2002), The Millennium Ecosystem Assessment (MEA, 2003) and The Economics of

Ecosystems and Biodiversity (TEEB, 2010b). These divide ecosystem functions and services into four main categories: regulating services, such as the regulation of climate, maintenance of soil fertility and waste-water treatment; habitat or supporting services, such as habitats for species and maintenance of genetic diversity; provisioning services, which include food, raw materials, water and medicinal resources; and cultural services, like recreation and aesthetic appreciation.

However, there is not yet agreement on the conceptualization of biodiversity and its relationship with ecosystem functioning and services. According to MEA (2005), biodiversity represents the foundation of ecosystems that, through the services they provide, affect human well-being. Thus biodiversity is an abstract notion that affects the generation of a multitude of ecosystem services, and is associated with notions like integrity, stability and resilience of complex systems.

Another view is expressed by Mace et al. (2012). They conceptualize biodiversity as (a) a regulator of ecosystem functioning, (b) a final ecosystem service and, (c) a good that has value of its own. The first role is argued to be the most important one. For example, the dynamics of many soil nutrient cycles are determined by the composition of biological communities in the soil while resilience to pests or environmental change improves with more diverse biological communities. The second characterisation responds to the argument that biological diversity at the level of genes and species contributes directly to some goods and their values. For instance, the potential value of wild medicines and the potential benefits from bioprospecting for medicinal purposes increase directly with the number and genetic distinctiveness of species. The third conceptualization follows from biodiversity being itself the direct object valued by humans. Many components of biodiversity may be seen to have cultural value, including appreciation of wildlife and scenic places and spiritual, educational and recreational values. Nevertheless, biodiversity is difficult to disentangle and measure which suggests that considering it directly as a service or a good to which instrumental value is assigned can be problematic. This is further discussed in Section 2.4 on biodiversity values.

In addition, there is debate on the role of biodiversity in delivering or enhancing ecosystem services provision. Some authors state that biodiversity can enhance ecosystem productivity (production of ecosystem services) and ecosystem stability. Generation of ecosystem services has been related to biological characteristics and more specifically to functional traits of ecosystems. Recent studies have argued that the multiple associations between functional traits and services, so-called trait-service clusters, can form the basis for ecosystem management and decision-making (Bello et al., 2010). For instance, for plants there is increasing evidence about the effects of community-level functional traits on ecosystem functioning that underlies important ecosystem services. A given ecosystem property could contribute to several ecosystem services: for example, diversity of flowering onset dates contributes to agronomic, cultural and pollination services (Lavorel et al., 2011). Furthermore, an ecosystem service is related to many ecosystem properties. For instance, high cultural value is related to high species diversity and highly diverse flowering characteristics. Some studies indicate that changes in biodiversity probably affect more regulating and cultural services, and long term resilience of ecosystem processes, and less provisioning services, at least in the short term (Mace et al., 2012; Lavorel et al., 2011).

In the specific case of agroecosystems, research suggests that their capacity to deliver a variety of ecosystem services depends on the intensity of use and on the diversity of croplands. For example, Sandhu et al. (2010) attribute a larger flow of ecosystem services to organic than to conventional agriculture, defined as agriculture based on monoculture and intensive use of agrochemicals, fuel, and machinery. In the same vein, Altieri (1999) and Jackson et al. (2007) argue that agriculture based on traditional practices like intercropping, agroforestry, or shifting cultivation delivers more ecosystem services than conventional agriculture, for various reasons. First, traditional agriculture largely relies on the maintenance of agrobiodiversity (Altieri, 1999; Jackson et al., 2007), thereby combining agricultural productivity with the delivery of other regulating services that biodiversity provides (MEA, 2005). Second, maintenance of agrobiodiversity in agricultural landscapes enhances the resilience of agroecosystems, i.e. their capacity to reorganize after disturbance, thereby enhancing the likelihood of maintaining the supply of ecosystem services over time in the face of variability and change (Jackson et al., 2007). Third, the adaptation of traditional agriculture to site-specific biological, edaphic, and climatic conditions reduces the dependence on inputs of machinery, agrochemicals, and fuel, thereby reducing related disservices in terms of soil compaction, water pollution, and greenhouse gas emissions (Altieri, 1999). For instance, maintenance of high biodiversity levels in specific taxonomic groups (maintenance of landraces) improves the performance of ecosystem services by enhancing pest control, pollination, or soil fertility (Altieri, 1999; Jackson et al., 2007). In addition, the habitat service “maintenance of landraces” is tightly connected with important cultural services, such as “heritage value of home gardens and associated traditional ecological knowledge” and “place for creating and enhancing social networks” since both landraces and knowledge are spread throughout seed exchange networks (Calvet-Mir et al., 2012).

As has been shown, there is growing consensus among ecologists that, in general, biologically diverse ecosystems provide a greater flow of ecosystem services than non-diverse systems (Hooper et al., 2005; Lavorel et al., 2011). Nevertheless, characterizing multiple ecosystem services and biodiversity across the same region has only recently emerged as a field of study, which means that levels of congruence are still poorly understood. The little quantitative evidence available to date has led to mixed conclusions (Chan et al., 2006). Comparing eco-region distribution data for biodiversity and a limited set of ecosystem services, Naidoo et al. (2008) find that optimizing for individual ecosystem services (carbon sequestration, carbon storage, grassland production of livestock, water provision) conserved only 22–35% of the species for a given area as did optimizing for species, that is, no more than were conserved by selecting ecoregions at random. They also found that maximizing species representation for a given area captured only 17–53% of maximum ecosystem service provision, depending on which service was considered and at which area limit the comparison was made. These levels of ecosystem service capture from species optimization were, again, no greater than those from a random selection of ecoregions (Naidoo et al., 2008).

Other studies exploring spatial patterns in the distribution of ecosystem services across landscapes analyse the spatial concordance between ecosystem services and biodiversity. They find that ecosystem services and biodiversity are interdependent (Egoh et al., 2008; Goldman and Tallis, 2009). However, there remains disagreement about whether spatial congruence of ecosystem services and biodiversity is rare or not, and what this implies for ecosystem management. Without knowledge about relationships between biodiversity and ecosystem services provision and among ecosystem services, we are at risk of designing policies that imply unwanted trade-offs.

2.4 Biodiversity and ecosystem services values

Valuation can be seen as the process of assigning importance to objects and actions. Pascual et al. (2010) mention two major types of valuation, namely (a) ecological valuation based on bio-physical accounting which neglects human needs or wants, and (b) economic valuation based upon consumer preferences. The latter takes the form of monetary valuation using market and non-market valuation approaches. In addition, one can identify socio-cultural valuation using a subjective evaluation approach (e.g., using a Likert scale) (Brondízio et al., 2010; Calvet-Mir et al., 2012). The relevance of group-based, social and cultural valuation in relation to biodiversity and ecosystem services has gained recognition (EPA-SAB, 2009).

Valuing biodiversity economically is controversial (Ring et al., 2010). An optimistic perspective is based on the idea that one is able to disentangle or decompose the total economic value of biodiversity into different types of values (as discussed in Box 2.2). The economic value of ecosystem services refers to instrumental values, resulting from the interaction of a human subject willing to pay for a (change in) an object (the ecosystem service), as opposed to intrinsic values in which case the subject plays no role. Most environmental economists consider that valuing biodiversity is a necessary step to make rational and accurate choices and trade-offs. Pavan Sukhdev, coordinator of the TEEB report, considers valuation in the broadest sense, including cultural and social approaches, as a key tool for conserving biodiversity: ‘‘lack of valuation is, we are discovering, an underlying cause for the observed degradation of ecosystems and the loss of biodiversity” (TEEB, 2008, p.4)’. Any decision or policy affecting biodiversity implicitly assigns a value to it. Moreover, despite its shortcomings, monetary valuation of welfare impacts – particularly when using a referendum type of format – might be considered as a democratic approach to decide about public policy regarding biodiversity, that is, as long as certain conditions are fulfilled, such as having a not too uneven income distribution and equal access to ecosystem services.

These are all arguments in favour of economic valuation of biodiversity with which one can agree or disagree. If one strives to support public policy with information about biodiversity values, then one needs a clear understanding of the relationships between biodiversity types, ecosystem functioning, ecosystem service categories, and biodiversity policies as these define the scenarios to be valued. Figure 2.1 already provided a schematic perspective on these relationships.

Decomposing the total value of biodiversity into direct and indirect use, non-use, option or quasi-option values as in Box 2.2 is difficult for a number of reasons (Nunes and van den Bergh, 2001). One is that there are different types or levels of diversity as discussed in Section 2.2, so the question is which one needs to be valued. In addition, valuation will lead to an under-estimation of the 'real' value because so many links between biodiversity and value categories are easily overlooked or simply cannot be empirically assessed. Some feel uncomfortable with putting an instrumental value on biodiversity and argue that biodiversity mainly has intrinsic value (Ehrenfeld, 1988). This view regards biodiversity as an abstract notion that is associated with notions like integrity, stability and resilience of complex systems, and thus difficult to disentangle and measure. One may see the value of ecosystems and their services as a metaphor, which is useful in communicating science-based insights to policy makers. The success of the much debated "Value of Nature" article by Costanza et al . (1997) can perhaps be understood in this way. Kosoy and Corbera (2010) point out that monetary valuation runs a risk of leading to partial or incomplete sets of values of ecosystems upon which policies and strategies will based, which then neglect non-monetized values.

All in all, it is unavoidable that there are different opinions on biodiversity value. In the United States, where executive orders often require economic cost-benefit analyses, the Environmental Protection Agency is now actively promoting the use of a wider range of valuation methods, including measures of attitudes, preferences and intentions, civic valuation, decision science approaches, ecosystem benefit indicators, and biophysical ranking methods (EPA-SAB, 2009; Ring et al., 2010). From a collective choice perspective, social norms and institutions are crucial for societal decision making (Vatn and Bromley, 1994). As alternatives, consensual, multi-criteria, multi-stakeholder and group-based deliberative valuation processes have been suggested as more appropriate. Here people act as citizens, not (only) as consumers (Funtowicz et al., 1998; Lienhoop and MacMillan, 2007; Spash, 2008a, 2008b; Spangenberg and Settele, 2010). Others feel that laypersons cannot judge the relevance and complexity of biodiversity-ecosystems-functions-services relationships and thus are unable to value biodiversity and associated ecosystem services appropriately. Instead, judgments about biodiversity changes are then better left to experts, like biologists. An intermediate solution is to let experts inform laypersons before confronting the latter with valuation questions (Nunes and van den Bergh, 2001). Surprising perhaps, Calvet-Mir et al. (2012) found a high statistically significant correlation between the responses by laypersons and a panel of scientists on the socio-cultural valuation of ecosystem services provided by home gardens. Finally, according to David Pearce (1999), "… much of the literature on the economic valuation of 'biodiversity' is actually about the value of biological resources and it is linked only tenuously to the value of diversity…" But whereas biodiversity refers to the variety of life, biological resources refer to the manifestation of that variety.

Box 2.2 A typology of economic values of biodiversity

Different types of economic value have been proposed in the literature (Nunes and van den Bergh, 2001; Pascual et al., 2010). *Utilitarian or (direct) use value* of components of biodiversity refers to the productive and consumptive uses of organisms or genes that are part of the local diversity as inputs into consumption and production processes. These are the subsistence and commercial benefits of species or their genes.

Indirect use value can be seen as the value of biodiversity contributing to ecosystem life support functions and the preservation of ecological structure and integrity (Swift et al., 2004). It can also denote biodiversity at a certain location affecting through complex ecosystem links a value at other locations. Barbier (1994) defines it as "… support and protection provided to economic activity by regulatory environmental services …" (p. 156). Different terms for the same notion are *contributory value, primary value,* and *infrastructure value* of biodiversity (see Farnworth et al., 1981, Norton 1986, Gren et al. 1994).

Option value is the value (a kind of use value) of keeping an option open for potential future use. *Quasi option value* is the value of being able to obtain information by keeping an option open, such as learning about unique species in the future by preserving all tropical forests. *Bequest value* represents the value of biodiversity for use by our offspring, or more generally future generations. *Philanthropic (altruist) value* is the value associated with use by others in our generation.

Non-use value is the value that biodiversity has on its own, without a (human) subject using it. According to some this value comprises cultural and social benefits, although the exact separation with use values is debatable (that is why some prefer the term *passive use value*). Indeed, use is often implicit, like in the case of watching movies or photos of species or nature. One has to distinguish here between *intrinsic* and *existence values.* The first is really another value concept (without a subject, so non-transformable into monetary units, but instead often taking the form of a "right"). *Existence value* of an environmental entity reflects humans capturing its intrinsic value or the instrumental value it has for ecosystems or nonhuman species (Attfield, 1998).

Table 2.1 illustrates possible connections between the most relevant biodiversity dimensions that contribute to the provision of each of the four types of ecosystem services and the most relevant economic values associated with these services.

Table 2.1 Illustrating the most relevant relationships between ecosystem services, biodiversity dimensions and economic values

Ecosystem service	Most relevant biodiversity dimensions	Most relevant economic value types
Regulating (climate regulation, waste water treatment, pollination, etc.)	Functional, ecosystem	Indirect use, option value
Habitat/supporting (habitats for species, maintenance of genetic diversity)	Genetic, functional	Indirect use, quasi-option, existence
Provisioning (water, raw materials, food, medicinal resources, etc.)	Genetic, taxonomic, functional	Direct use, option, quasi-option, bequest
Cultural (knowledge, recreation, aesthetic, etc.)	Taxonomic, ecosystem	Direct use, option, existence, bequest

2.5 A typology of rebound of biodiversity policies

After exploring the main definitions of biodiversity in the literature and establishing the link between biodiversity, ecosystem services and their values to society, in this section we aim to characterize the potential rebound effects of biodiversity policies. This will allow us to advance in the promotion of more effective policies. A biodiversity policy can be considered to be effective if it will produce the conservation benefits as desired ex ante (Doremus, 2003). One might also define effectiveness in a more abstract way as attaining the highest marginal environmental benefit associated with a given instrument (OECD, 2007; Ring and Schröter-Schlaack, 2011).

Different biodiversity conservation policies can deal with particular causes of biodiversity loss, such as hunting, habitat destruction (land use, deforestation, fragmentation), water use (causing desiccation), and environmental pollution (with climate change as a special and very important case). At a more fundamental level, one can identify environmental externalities, myopia (a high rate of time discounting), a lack of adequate property rights, and to a lesser extent market power and asymmetrical information, as the indirect causes of biodiversity loss. Different types of biodiversity conservation instruments can be designed to deal with these various causes. On the one hand, policies may aim to provide prohibitions, barriers, standards (e.g., land tenure and use rights), or negative incentives like prices (subsidies, land or product taxes, access fees) to alter behaviour and projects that negatively affects biodiversity (TEEB, 2010a, 2011). The most important biodiversity policy instruments are summarized in Table 2.2.

Table 2.2 Biodiversity conservation instruments and their characteristics

Instruments	**Incentive**	**Direct target**
	Regulatory instruments	
Direct regulation and spatial planning	Coercion	Various behaviours that negatively affect biodiversity
	Economic instruments	
Biodiversity offsets and mitigation banking	Avoiding a fine	Planning of projects that harm biodiversity
Environmental taxes	Tax	Various behaviours that negatively affect biodiversity
Tax reliefs	Avoiding a tax	Various behaviours positively affect biodiversity
Ecological fiscal transfers	Payment	Various behaviours that positively affect biodiversity
Environmental subsidies	Payment	Various behaviours that positively affect biodiversity
Government financed payments for environmental services	Payment, contract	Compliance with terms of contract to protect biodiversity
Market-based payments for environmental services	Payment, contract	Compliance with terms of contract to protect biodiversity
	Voluntary and information-based instruments	
Voluntary instruments	Prevention of coercive regulation	Compliance with voluntary agreement or pledge to protect biodiversity
Certification	Regulating access to a market or gaining a good reputation	Compliance with code of conduct to protect biodiversity

Source: adapted from Ring and Schröter-Schlaack (2011).

The types of policies in the table are likely to score differently in terms of effectiveness, depending on the context and application. This section aims to draw attention to a kind of government or policy failure that affects effectiveness, namely the unintended indirect effects and potential ineffectiveness of policies. Biodiversity policies can have a number of unintended, unwanted and avoidable rebound effects. If they would not be avoidable it would not make sense to bother about them, although one could then perhaps think of compensation measures.[8] We propose the following typology of rebound:

[8] We are not assuming more than 100% rebound here. The latter is also known as the "Jevons paradox" in the context of energy rebound, because of the English economist William Stanley Jevons who in his 1865 book "The Coal Question" drew attention to the risk that a more efficient steam engine would increase rather than decrease the demand for coal. Of course, more than 100% biodiversity rebound should withhold one from implementing the respective biodiversity conservation policy in the first place.

1. *Biodiversity rebound I (spatial spillover)*: Policy to protect one type of biodiversity in a certain area has a negative impact on such biodiversity elsewhere, i.e. in another region. This rebound operates through spatial spill-over effects which some have called displacement or leakage (Lambin and Meyfroidt, 2011). An example is restricting outdoor recreation in one nature area that leads to recreationists moving to other areas so that environmental pressure there increases with potentially negative impacts on biodiversity. Or deviating water flows in the landscape to assist in the protection of biodiversity in a wetland can lead to water shortage and desiccation in other nature areas with consequences for respective biodiversity. Evaluations of the effectiveness of protected areas showed that these conservation policies may lead to an increase in deforestation rates outdoors of those protected areas (Lambin and Meyfroidt, 2011) reducing habitats for biodiversity conservation. In the case of Sumatra, a reduction of deforestation in adjacent unprotected areas was observed, probably due to urban migrations (Gaveau et al., 2009). By contrast, a study in the Peruvian Amazon (Oliveira et al., 2007) found that although forest subject to legal concessions experienced a large reduction in deforestation, after enactment of stringent timber harvest legislation the rates of forest clearing and disturbance outside areas with concessions increased rapidly. Protection of public forests in the US Pacific Northwest also displaced timber harvests on private timberlands in the region and further away, with a total displacement of 84% of the reduced public harvest timber because of conservation programs (Wear and Murray, 2004). A similar leakage effect was found for cropland in the United States, where the purchase of conservation easements on farmland brought non-cropland into crop production elsewhere, for about 20% of the cropland area that was retired from cultivation (Wu, 2000).[9]

2. *Biodiversity rebound II (incongruence between protection of different types of biodiversity)*: Policy to protect one type of biodiversity (e.g., genetic) can negatively affect another type of biodiversity (e.g., taxonomic or functional). As has been discussed in the previous sections, areas of high conservation interest are traditionally defined as biodiversity hotspots, but sometimes they are based upon rather arbitrary criteria. In fact, both past and current conservation strategies have frequently focused on giving priority to certain taxa or areas to protect rarity, endemism and distinctiveness (Hooper et al., 2005). For instance, French protected areas have underrepresented functional diversity, having been established following

[9] We do not claim that all biodiversity policy is subject to rebound. For instance, Andam et al. (2008) find in a study for Costa Rica that deforestation spillovers from protected to unprotected forests are negligible. Our aim here is merely to classify potential channels or mechanisms of rebound of biodiversity policies.

taxonomic diversity patterns (Devictor et al., 2010). Another example is providing incentives for habitat protection through creating corridors between protected areas which may increase disease risks by promoting contact between wild and domesticated animals (Kinzig et al., 2011). Only if all types of biodiversity are perfectly correlated will protection of one imply protection of the others, so that there are no conflicts and trade-offs required. However, as indicated by the conclusions of Section 2.2, this is unlikely to be the case.

3. *Ecological rebound:* As has been shown in Section 2.2, changes in biodiversity may lead to various responses in ecosystem functioning, some intended and foreseen but others not. As a result, biodiversity conservation policy might through its effect on particular biodiversity work out negatively on certain ecological relations. For example, red-list species conservation schemes can lead to population growth of particular species, in turn giving rise to a loss of equilibrium between different species in the ecosystem, because of food scarcity or predator pressure. This is discussed in more detail in the illustration of the Weitzman assessment of biodiversity policy below. When ecological changes affect ecosystem functional diversity, this rebound type overlaps with biodiversity rebound II.

4. *Service rebound (trade-off between biodiversity and ecosystem services)*: Although biodiversity conservation policy is increasingly justified based on the ecosystem services provided, there is still incomplete empirical evidence that there exists a strong relationship between biodiversity conservation and supply of ecosystem services. In fact, biodiversity policy may protect a certain type of biodiversity while degrading or sacrificing a particular ecosystem service. We recognize here that two different types of trade-offs exist, one between ecosystem service provision and biodiversity and another between different types of ecosystem services (Raudsepp-Heame et al., 2010; Ring et al., 2010; Willemen et al., 2010). Since we are concerned with biodiversity conservation the first type of trade-off is more relevant here. An example is the case in which biodiversity conservation implies a transformation of landscapes formed by a combination of culture and nature to more pure nature, with a loss of cultural values as a result. Another example is provided by the study done by Chan et al. (2006). It examines the potential trade-offs between goals for biodiversity and for certain ecosystem services. The authors find that there is a low average correlation between biodiversity and the six services studied (carbon storage, flood control, forage production, pollination, recreation and water provision). Moreover, crop pollination and forage production show a negative correlation with biodiversity. Another case of a trade-off between biodiversity and ecosystem services is conserving certain species that need dense, old-growth or primary forests, such as the boreal owl (*Aegolius funereus*), and provisioning ecosystem services, like grazing and timber production. An example of service

rebound in the context of marine ecosystems is a protected area policy aimed at forbidding fishing in order to conserve certain protected species. This can reduce provisioning services for fishermen while increasing biodiversity in the protected zone, although there are examples where marine protected areas may considerably increase fish catches close to their edges due to regenerating fish stocks within the protected areas (Flogarty and Botsford, 2007). There is a fundamental incongruence, and thus a conflict and need for trade-off, between (maintenance of) biodiversity and ecosystem services. As a result, this type of rebound can never be completely removed. Therefore, it is likely that in case of provisioning services such a trade-off will lead to increased demands on services in other locations in order to fulfil the (worldwide) demand for them. Such displacement of service demand (some- what resembling biodiversity rebound I: spatial spillover) may negatively affect ecosystems elsewhere. In case of services that are more location specific such displacement is less likely.

5. *Environmental rebound*: Biodiversity policy can generate a negative impact on certain environmental indicators. This is also known in the literature as shifting or cascading (Lambin and Meyfroidt, 2011) of one to another environmental problem. For example, biodiversity conservation leading to less use of tropical hardwood may lead to a shift in consumption and associated industries to other construction materials that involve chemicals or toxic components, or use a lot of CO_2-intensive energy. This then means a shift to other types of environmental problems. This will not always be easy to empirically demonstrate, as it involves 'invisible' behavioural and economic mechanisms. Harvey et al. (2010) mention carbon leakage as a potential risk of REDD (Reducing Emission by Deforestation and Degradation) aimed at a combination of carbon sequestration and biodiversity protection.

Note that rebound can occur through physical processes and displacement: e.g., using water to maintain wetlands may create drought conditions elsewhere (e.g., on a river trajectory) and put pressure there on biodiversity. Alternatively, rebound can involve an economic mechanism: e.g., spending limited budget on biodiversity protection in one spot may lead to deviating money from conservation elsewhere, or behavioural change stimulated by conservation policy (e.g., through environmental taxes or ecolabels) leads to new consumption and production activities that cause pressure on biodiversity or other environmental media. In addition, rebound can be local or nearby (like when water use affects adjacent ecosystems) or distant in space (because of economic or large-scale environmental processes).

Two other, related aspects of biodiversity loss and conservation matter for rebound of particular policies: the combined effects of multiple factors and pressures behind biodiversity loss, and the interaction or synergy of multiple, simultaneously active policies (policy mix) (Schröter-Schlaack and Ring, 2011). This complex nature of the interaction of causes and policies should be addressed when one aims to completely assess the potential rebound of biodiversity policy.

Finally, rebound also can result from global agreements on biodiversity that have all kinds of local effects, some of which are unintended. Such agreements need to develop effective mechanisms to eliminate such rebound effects to make the policies more effective. For instance, international policies that express support for conservation, such as the Convention on Biological Diversity, typically fail to have adequate precision and clarity to save many of the unique, agrobiodiversity-rich areas on the planet (Harrop, 2007). Instead, village level and regional institutions often may assure more biodiversity conservation through the engagement of local communities in activities that improve their livelihoods (Bawa et al., 2007; Jackson, 2007).

Weitzman on biodiversity policy: genetic distinctiveness and ecological rebound

Systematic conservation planning has traditionally focused on identifying priority areas that ensure adequate representation of measures of taxonomic diversity, such as species richness (Margules and Pressey, 2000). Consequently, as has been shown in many studies, functional diversity has been significantly under-represented whereas taxonomic diversity has been significantly over represented in protected areas (Devictor et al., 2010). Also in economic models of biodiversity loss, biodiversity is mainly considered at the species level, paying attention to taxonomic diversity, or in some cases prioritizing species conservation based in genetic information (Eppink and van den Bergh, 2007).

To illustrate potential ecological rebound (rebound type 3) of policies that focus on specific biodiversity, consider the approach of Weitzman (1998). He studied the problem of protecting biodiversity under a limited budget constraint, but without considering certain ecological dynamics. It is relevant here as it is a well-known approach that is regarded by many economists as useful for biodiversity policy assessment.

He derives the following criterion for setting priorities among biodiversity-protecting projects:

$$R_i = (D_i + U_i)\, {\Delta P_i}/{C_i} \qquad (1)$$

Here, R_i represents the performance index of species *i*, D_i is the (genetic) distinctiveness of species *i* (meaning roughly how unique or different a species is), U_i denotes the direct utility associated with preservation of species *i*, and C_i is the cost of the protection project that increases the probability of survival of species *i* by ΔP_i. Uncertainty of extinction is introduced by defining P_i as the probability of survival of species *i*, so that $1 - P_i$ is the probability of extinction of species *i*. These probabilities are exogenous, i.e. they originate from outside Weitzman's framework.

Van der Heide et al. (2005) draw attention to the lack of ecological considerations in Weitzman's criterion. They suggest that the ecological interdependence among species can in the context of Weitzman's criterion be modelled by defining mutually dependent rather than exogenous, independent survival probabilities. For their survival species depend very much on other species, through food web and ecosystem relationships. This implies that, generally, the extinction of one species will have an impact on the survival probabilities of certain other species. The conclusion is that Weitzman's ranking criterion generally holds only under very limited conditions: namely, when the probabilities of extinction of species are exogenous and constant. This assumption seems to hold mainly, and perhaps only, for ex situ conservation, which severely limits application of the criterion. Applying Weitzman's criterion to in situ conservation can provide an incorrect ranking of biodiversity policies leading to ecological rebound because it misunderstands ecological relationships between species. Note that this in turn means that biodiversity rebound is relevant here, as ecological (species) relationships determine functional diversity.

2.6 Concluding remarks

We have presented a framework of relationships between biodiversity, ecosystem functioning, ecosystem services and various types of rebound of biodiversity policy. The concern behind identifying potential rebound mechanisms is to design effective policies for biodiversity protection. Making sure that biodiversity policy is effective is a necessary condition for realizing a transition to a truly sustainable economy.

We have provided a preliminary classification into five types of rebound (biodiversity-spatial, biodiversity-incongruence, ecological, environmental and service) and have provided a preliminary set of illustrations of these rebound mechanisms. Some types of rebound relate to conflicts and the need for trade-offs between different types of biodiversity, or between certain types of biodiversity and certain ecosystem services. We do not claim any definite results, but merely offer a starting point for research. We hypothesize that including rebound effects in the analysis of biodiversity will alter policy conclusions.

Which particular research approach is needed to study these various types of rebound? It will require close collaboration between natural and social scientists, a good understanding of the various direct and indirect (fundamental) causes of biodiversity loss, a clear choice of relevant biodiversity measures, and a translation of past research in clear conclusions about connections between biodiversity, ecosystem functioning and services. One very likely will need to use systems models for concrete cases to assess all the unwanted and avoidable rebound effects of particular policies of biodiversity protection. In addition, it is useful to examine which policies, and in which settings, are functioning relatively well in terms of generating a low rebound and thus having a high effectiveness. Against this background, it would seem useful to connect the instruments in Table 2.2 to the rebound typology. This involves further conceptual thinking along the lines as sketched here, as well as studying different cases and ecosystems to understand the relevance of particular combinations of instruments and contextual factors for the magnitude of rebound.

Chapter 3. Sociocultural valuation of ecosystem services to improve protected area management: a multi-method approach[10]

3.1 Introduction

Around the world, cultural landscapes have developed as a result of intense interaction between human societies and biophysical systems (Plieninger and Bieling, 2013). This is the case of some regions in the Mediterranean area (Blondel, 2006; Farina et al., 2003). Here the maintenance of socio-ecological systems often depends on traditional rural activities by peasants, such as agro-silvo-pastoral practices (Otero et al., 2013). Understanding the interaction between human preferences, activities and landscapes is important for designing effective and socially acceptable nature management regimes, notably in protected areas that exist in many countries. Here we focus on Natural Parks which are a common instrument of such management. They correspond to category V, i.e. "Protected Landscape", of the IUCN classification of protected areas, reflecting the ecological, biological, cultural and scenic value of these areas (IUCN, 1994).

Mediterranean protected landscapes (e.g. natural parks) are under considerable pressure, especially when close to urbanized areas. There is not only intense land-use and land degradation in surrounding areas (Hanssen and DeFries, 2007), but also fewer traditional activities like agriculture and associated land abandonment (Plieninger et al., 2013). Moreover, there are new land uses appearing, notably leisure activities. In fact, according to Buijs et al. (2006) the increased appreciation of landscapes as leisure spaces might turn them into merely a décor for spending free time.

The noted pressures and new land uses may influence social perceptions of cultural landscapes and values people attribute to them. Understanding better such perceptions – including conflicting ones between individuals – can contribute to a better natural park management. Social perceptions and values can be assessed through a socio-cultural valuation of ecosystem services. The latter denote the direct or indirect contributions of ecosystems to human benefits or satisfaction of needs (MEA, 2003). In this chapter we report a socio-cultural valuation of (place-based) ecosystem services provided by the natural park of Sant Llorenç del Munt i l'Obac (hereafter Sant

[10] This chapter also appears as: Maestre-Andrés, S., Calvet-Mir, L., van den Bergh, J.C.J.M. 2016. Sociocultural valuation of ecosystem services to improve protected area management: a multi-method approach applied to Catalonia, Spain. *Regional Environmental Change* 16:717-731 Doi 10.1007/s10113-015-0784-3.

Llorenç") in Catalonia, Spain. This represents a relevant case of a Mediterranean protected landscape. Our approach involves four methods – (1) a literature review, (2) non-participant observation, (3) semi-structured interviews and (4) a valuation survey.

Pascual et al. (2010) mention two different valuation approaches, namely a) biophysical valuation which derives values from measurements of the physical costs (in terms of labour, energy or material inputs) of maintaining a given ecological state; and b) economic valuation through eliciting human preferences using market and non-market techniques of monetary valuation. A third valuation approach is socio-cultural valuation. It explores human attitudes and perceptions regarding ecosystem services for human well-being through (non-monetary) ranking methods (Martín-López et al., 2014; Maestre-Andrés et al., 2012). It is aimed to capture the multidimensional nature of value when referring to ecosystems (Martín-López et al., 2014; Kumar and Kumar, 2008), including less tangible social and ethical concerns (associated with non-material benefits) of ecosystems (Chan et al., 2012a, 2012b) The idea behind this approach was suggested some time ago already (e.g., De Groot et al., 2002), but still has seen few concrete applications (exceptions are Martín-López et al., 2012; Calvet-Mir et al., 2012; Oteros-Rozas et al., 2014). We employ a socio-cultural valuation because we are interested in assessing the system as a whole rather than (small) changes in it. For the latter, one might also use monetary valuation approaches. If the value of the change in a system is too large in comparison with the income of the individual whose preferences for the change are elicited monetary valuation is less suitable and reliable. In the case of socio-cultural valuation larger changes can be studied as there no such income constraint holds. As monetary and biophysical valuation have already seen many applications to ecosystem services (Vihervaara et al., 2010; Nieto-Romero et al., 2014), it is also useful to undertake more socio-cultural valuation studies so as to add new perspectives and diversity of insights. It has been used to inform landscape management and planning about stakeholder needs and values and to identify potential conflicting views between stakeholders (Castro et al., 2011; Agbenyega et al., 2009; Casado-Arzuaga et al., 2013; Pereira et al., 2005). This is particularly relevant in protected areas where multiple interest groups may have different priorities concerning the protection or improvement of ecosystem services. Only few studies have assessed socio-cultural perceptions of ecosystem services by stakeholders in protected areas, which will be the focus of our study (Martín-López et al., 2012; and Sodhi et al., 2010). Stakeholder involvement in identifying and defining ecosystem services (in cooperation with natural and social scientists) permits to understand local people's perception of the contribution of nature to their well-being (Menzel and Teng, 2009; TEEB, 2010b).

In view of this approach, the specific goals of this chapter are: 1) to examine the understanding of the term ecosystem services by stakeholders and visitors of the natural park of Sant Llorenç; 2) to identify and characterize the ecosystem services provided by the natural park by stakeholders; 3) to elicit the perceived importance by visitors of different categories of ecosystem services and particular ecosystem services; 4) to identify the factors that influence people's valuation of ecosystem services; and 5) to detect conflicting views among stakeholders reflected by diverging socio-cultural preferences regarding the park and the ecosystem services it provides.

The remainder of the chapter is structured as follows. Section 3.2 briefly discusses the four methods of the study. Section 3.3 presents the results, which are then discussed in Section 3.4. Conclusions and general lessons are drawn in Section 3.5.

3.2 Methods

Research was conducted in the natural park of Sant Llorenç (Figure 1.1) and its surroundings between January and November 2013. Our research involved a combination of qualitative and quantitative methods: 1) a literature review of ecosystem services; 2) non-participant observation; 3) semi-structured interviews (N=25) among relevant stakeholders of the natural park; and 4) a survey (N=200) among visitors of the natural park. Table 3.1 shows the links between the research goals defined in Section 3.1 and the methods used to tackle them. The first three methods are used in a preliminary phase to provide essential input for designing the fourth one; in addition, methods 3 and 4 provide complementary information for arriving at insights about perceptions and values of ecosystem services, while 2 and 3 jointly identify conflicting viewpoints on park management. Based on the assessed perceptions, values and conflicting viewpoints we can then suggest improvements in park management.
All these methods are explained below.

Table 3.1 Connection between research goals and methods of data collection

Methods	**Research goals**				
	1. assess the understanding of the term ecosystem services	2. identify and characterize ecosystem services	3. determine the perceived importance of ecosystem services	4. assess factors that influence socio-cultural valuation	5. identify (potential) conflicting views among stakeholders
A review of the literature on ecosystem services		●			
Non-participant observation		●			●
Semi-structured interviews with stakeholders	●	●			●
Survey of visitors to the park	●		●	●	

3.2.1 Methods of data collection

3.2.1.1 Literature review

In order to identify and characterize the ecosystem services of the natural park, potential ecosystem services were distinguished through a review of the relevant literature. Following the classifications of de Groot et al. (2002), The Millennium Ecosystem Assessment (MEA, 2003) and The Economics of Ecosystems and Biodiversity (TEEB, 2010b) we divided ecosystem services into four main categories: regulating, habitat, provisioning and cultural. We avoid using the category supporting services established by The Millennium Ecosystem Assessment since it may lead to double counting in valuation practices (Hein et al., 2006).

3.2.1.2 Non-participant observation

We used non-participant observation techniques to establish contact with the community, local culture and local social organization in a non-active way (Bessette, 2004). The community comprises the people living inside the natural park and in the municipalities whose areas are partly located within the borders of the protected area. The non-participant techniques used included several activities by the researchers, including trips in the period January to June 2013 to obtain a basic knowledge about the natural park. This permitted us to identify the potential ecosystem services that can be provided by the natural park and comprehend conflicting views between different stakeholders within the park. During these trips we visited the twelve municipalities whose areas are wholly or partly located within the borders of the natural park and had informal talks with individuals or groups in order to know the relationship of each municipality with the natural park. For example, we spoke with inhabitants (people in bars, in squares and so on) about their sense of place regarding the natural park. In addition, we stayed two weeks in July and August 2013 in a farmhouse inside the park. This stay gave us the opportunity to communicate with farmers and people living inside the protected area and get to know their characteristics and their associated conflicting views with other stakeholders and visitors. We participated in a meeting of the Advisory Committee (i.e. an informative meeting of stakeholders and park managers held every six months and open to everybody) in order to observe how stakeholders interact and to identify conflicting views among them.

3.2.1.3 Semi-structured interviews

We conducted 25 interviews with relevant stakeholders of the natural park from January to June 2013. We selected stakeholders who had an interest in ecosystem services because they benefited from them or had an influence on their provision (Reed et al., 2009). They were selected on the basis of reputation or recommendation (following a "snowball strategy"). We distinguished 9 different categories of stakeholders corresponding to sectors present in the natural park such as local administrations (mayor and councillors of environment), park managers and employees, representatives of conservationist organisations, workers in the agricultural, scientific, tourism, leisure, education, and forestry sectors. The interviews lasted 1.5 to 2 hours and were all recorded with previous consent. The interviews were structured in sections dealing with the following topics:

1) Whether stakeholders were familiar with the term ecosystem services. A brief explanation of the meaning of the concept of ecosystem services was provided to the people lacking knowledge about it.
2) Place-based ecosystem services provided by the natural park as perceived by stakeholders.
3) Ecosystem services that can be provided with appropriate ecosystem management as perceived by stakeholders.
4) Conflicting views among stakeholders in the natural park.

3.2.1.4 Survey

We designed a survey drawing on the information about identification and characterization of ecosystem services in the previous step. To test the questionnaire we undertook 15 pilot surveys. We conducted the survey among 203 visitors of the natural park during the months of July and August 2013. Three questionnaires were not entirely completely filled in, so that we ended up with a sample size of 200. The sample of visitors was randomly selected and was restricted to individuals over 18 years old. The surveys were done at seven different sites of the natural park. The survey was structured in two main sections. In the first section we used a Likert scale design (Bernard, 2005) to assess visitor's agreement on statements about the importance of the ecosystem services of the natural park. The level of agreement with the statements followed a scale from zero to five, with zero denoting "I completely disagree" and five "I completely agree". We presented to visitors a statement referring to each of the 28 ecosystem services previously identified and characterized. In order to facilitate the interpretation, each ecosystem service was presented using an illustrative photograph in the context of the natural park of Sant Llorenç. For example, when we introduced to the respondent the statement "the natural park is important because it contributes to pollination" we showed the visitor an illustrative photograph of bees in Sant Llorenç. Then we asked the respondent his or her level of agreement. They also had the option to not value it. In the second section, we collected information about socio-demographic characteristics and environmental behaviour of the surveyed individuals (e.g., age, sex, formal education, consumption of products from organic agriculture). While conducting the surveys, the term ecosystem service was always referred to as "the benefits that natural park provided for human well-being" to make the term more understandable. At the end of the survey, we asked the respondents if they knew the term ecosystem services, and if not, how they would interpret it.

3.2.2 Methods of data analysis

In order to assess the understanding of the term ecosystem services (objective 1) we relied on data from interviews and surveys. We examined whether responses contained terms that were identical or similar in meaning to core terms in various accepted definitions of ecosystem services in the literature as discussed in the previous section. From the set of responses obtained, we distinguished between two different meanings attached by people to the concept of ecosystem services: actions done by humans to the benefit of nature (not in line with the common use in the literature), and the benefits provided by nature for human well-being (generally accepted meaning). We used descriptive statistics to assess the number of people (both stakeholders and visitors) who knew the concept.

With the aim of identifying and characterizing the place-based ecosystem services of the natural park of Sant Llorenç (objective 2), we triangulated the information obtained by 1) the literature review, 2) the non-participant observation and 3) the semi-structured interviews. We mean by characterization the process of attributing to each ecosystem service the significance given by stakeholders in the context of the natural park. For analyzing the interviews, a coding process was applied following Charmaz (2006) to categorize the information into different ecosystem services. We further categorized the ecosystem services that could be improved by better management. Once the preliminary list of potential ecosystem services was obtained, we used the information from the interviews to verify and expand our preliminary list obtaining the final list of 28 ecosystem services. We classified sources of information as 1) "literature" when the source of identification was literature review, 2) "observation" when the ecosystem service was identified by non-participant observation, and 3) "interviews" when the source of identification was semi-structured interviews.

In order to assess the importance attributed by visitors to the ecosystem services provided by the natural park (objective 3), we conducted a socio-cultural valuation through a survey among visitors. We calculated the average values for all ecosystem service and the average of these for each category (provisioning, regulating, habitat and cultural) based on the individual values obtained in the surveys.

We used the information from the surveys to assess which were the factors that influenced people's perception of the importance of ecosystem services (objective 4). First, we obtained the socioeconomic and environmental profile of the visitors of the natural park by means of descriptive statistics. Afterwards we generated the variable MeanES through the aggregation of all the scores given to each ecosystem service. For doing so, we first ensured that all answers in the survey measured the same construct using the coefficient of internal consistency Chronbach alpha (alpha=0.89). We selected 10 explanatory variables for statistical multivariate analysis, as shown in Table A3.1 in the Appendix. We had two continuous variables and 8 binary variables. To examine the association between the socio-demographic variables of visitors and the valuation of ecosystem services we run various ordinary least-square multiple regressions with MeanES and the scores for each of the 28 ecosystem services as dependent variables. For the statistical analysis we used STATA 12. In total, we thus did 1+28 =29 regressions (for the average and each of the 28 ecosystem services).

To test robustness of our results, we did an "outlier analysis". This analysis was based on selecting a sub-sample composed by the respondents who valued the ecosystem services the least and most, with pre-defined low and high cut-off points of the variable MeanES. The following procedure was used for this. The aim was to arrive at a sub-sample that was approximately 25% of the sample (i.e. about 50 outliers of a total of 200 respondents). To achieve this, we set the low cut-off point at 3.75 and the high one at 4.80, to make sure that the numbers of positive and negative outliers were close. This resulted in 25 low outliers and 28 high outliers, that is, 53 outliers in total. Next, we undertook the same multiple regressions for outliers as we described above for the whole sample. So in total, we performed 2x29=58 regressions (i.e. for the whole sample and the outliers).

We used the qualitative information obtained through semi-structured interviews and non-participant observation to identify conflicting views among stakeholders (objective 5). We coded the data gathered by using codes derived from the responses to the open questions (Newing, 2011).

3.3 Results

3.3.1 Understanding of the term "ecosystem services"

The term "ecosystem services" was unknown to most of the interviewees (N=19, 76%), except for 6 persons (24%) who were two park managers, one worker of the natural park, two scientists, and one person from the conservationist organisation. All stakeholders interviewed were aware of the wide range of benefits that the natural park provides. Nevertheless, they were confused to hear the word "service" when talking about nature because they attributed that term to human activity. In the case of the visitors surveyed, only 2.5% knew the meaning of the term ecosystem services. Subsequently, they were asked what would be the meaning they would attribute to the term. Only 13% provided a reasonably correct definition, 67.5% answered that it would be actions done by humans in order to improve nature, and 19.5% were incapable of giving any definition.

3.3.2 Identification and characterization of the 28 ecosystem services

The ecosystem services most noted in the interviews with stakeholders were habitat and cultural ones. In relation to the habitat category, 19 persons (76%) mentioned "gene pool protection" and 15 (60%) "lifecycle maintenance". Regarding the cultural category, 22 stakeholders (88%) mentioned "aesthetic information"; 17 (68%) identified “distraction and leisure” and 15 stakeholders (60%) mentioned “eco-tourism”, “spiritual experience” and “local identity and cultural heritage”.

As shown in Table 3.2, we identified and characterized 28 place-based ecosystem services: 8 regulating services, 2 habitat services, 6 provisioning services and 12 cultural services. Fifteen ecosystem services (53.6%) were identified through the three sources of identification; three (10.71%) through observation and interviews to relevant stakeholders; and six (21.42%) through literature review and interviews. Two services (7.14%) were identified only by the literature review and two (7.14%) only through the interviews. It is important to highlight that five place-based ecosystem services (17.86%) identified in the interviews or non-participant observation are not specifically mentioned (somewhat in line with their place-based nature) in the general literature on ecosystem services reviewed in Section 3.2.1.1. Table 3.2 also lists the characterization of each ecosystem service by stakeholders. For instance, the ecosystem service “disturbance prevention” was characterized as the prevention or moderation of floods.

Ecosystem services that can be improved or provided by better management were also identified. A considerable number of stakeholders (n=13, 52%), responded that “eco-tourism” should be properly managed and that tourists should be redirected to surrounding villages to generate economic benefits. Many stakeholders were of the opinion that the amount of people visiting the park should be reduced because it generates too much pressure on ecosystems (n=17, 68%). Ten stakeholders (40%) suggested that “food cultivation” should be enhanced, for a number of reasons: to ensure economic feasibility of living in the park, to maintain open areas that enhance biodiversity and diminish the risk of fires, to enhance agrobiodiversity through the cultivation of local varieties, to maintain traditional knowledge, and to recuperate farms that were abandoned. “Maintenance of traditional knowledge” was mentioned by six stakeholders (24%) as an ecosystem service that should be enhanced. “Raw materials” (n=6, 24%), especially biomass, was mentioned as a potential economic activity and a tool to enhance open areas that support biodiversity and diminish the risk of fires.

Table 3.2 Identification and characterization of the 28 ecosystem services provided by the natural park of Sant Llorenç

Ecosystem services	Characterization by stakeholders	Source of identification: Literature	Source of identification: Observation	Source of identification: Interviews
Regulating services				
Air purification	Provision of clean air	●	●	●
Climate regulation	Capture of CO_2 and mitigation of climate change	●	●	●
Disturbance prevention	Prevention or moderation of floods	●		●
Regulation of water flows	Water storage capacity due to vegetation cover	●	●	●
Waste treatment	Improvement of water quality	●	●	●
Erosion prevention	Control of soil erosion	●	●	●
Pollination	Contribution to pollination	●		
Biological control	Presence of biodiversity that prevents or moderate pests and diseases	●		
Habitat services				
Gene pool protection	Biodiversity conservation in Mediterranean ecosystem, unique species and local landraces	●	●	●
Lifecycle maintenance	Connection between natural areas	●	●	●
Provisioning services				
Food gathering	Provision of mushrooms and wild fruits			●
Food cultivation	Provision of vegetables, cereals and meat	●	●	●
Water	Provision of water	●	●	●
Raw materials	Provision of wood and fodder	●		●
Medicinal resources	Provision of various medicinal plants	●		●
Ornamental resources	Provision of decorative objects like flowers, wild pig heads, etc.	●		●
Cultural services				
(Eco-) tourism	To be a touristic place	●	●	●
Distraction and leisure	Offer of places for enjoying and spending free time	●		●
Physical recreation	Offer of opportunities for practicing different sports and keeping fit		●	●
Mental recreation	Contribution to disconnect, relax and diminish the stress			●
Hunting	Provision of wild animals for hunting		●	●
Aesthetic information	Provision of unique and attractive landscapes	●	●	●
Inspiration for culture, art and design	Contribution to artistic inspiration	●		●
Spiritual experience	Contribution to a direct connection with nature	●	●	●
Information for cognitive development	Research on and education about nature	●	●	●
Maintenance of traditional knowledge	Maintenance and exposure of traditional countryside activities and skills	●	●	●
Maintenance of social relations	To be a space where you can maintain or create social relationships among people and family	●	●	●
Local identity and cultural heritage	To be part of our personal and common history		●	●

3.3.3 Valuation of ecosystem services provided by the natural park of Sant Llorenç

Figure 3.2 presents the average value assigned to each ecosystem service (on a scale from zero to five). Of a total of 28 ecosystem services, 22 (78.57%) obtained an average score between 4 and 5, which indicates that the surveyed individuals perceive these services as very important. Four ecosystem services (14.29%) had an average value between 3 and 4, and only two (7.14%) between 2 and 3. None had an average value under 2. The ecosystem service most valued was “spiritual experience”. It was followed by “information for cognitive development”, “mental recreation”, “aesthetic information”, “physical recreation (sport)”, “gene pool protection” and “distraction and leisure”. All of these ecosystem services obtained an average value above 4.80 and are indicated by boxes in Figure 3.2. Note that all these, except “gene pool protection”, were cultural services. The least valued ecosystem services were “hunting”, which obtained an average score of 2.07; and “ornamental resources”, with a score of 2.79. The ecosystem services that most people found difficult to value due to unfamiliarity with their meaning were “disturbance prevention” (n=16, 8%) and “biological control” (n=11, 5.5%).

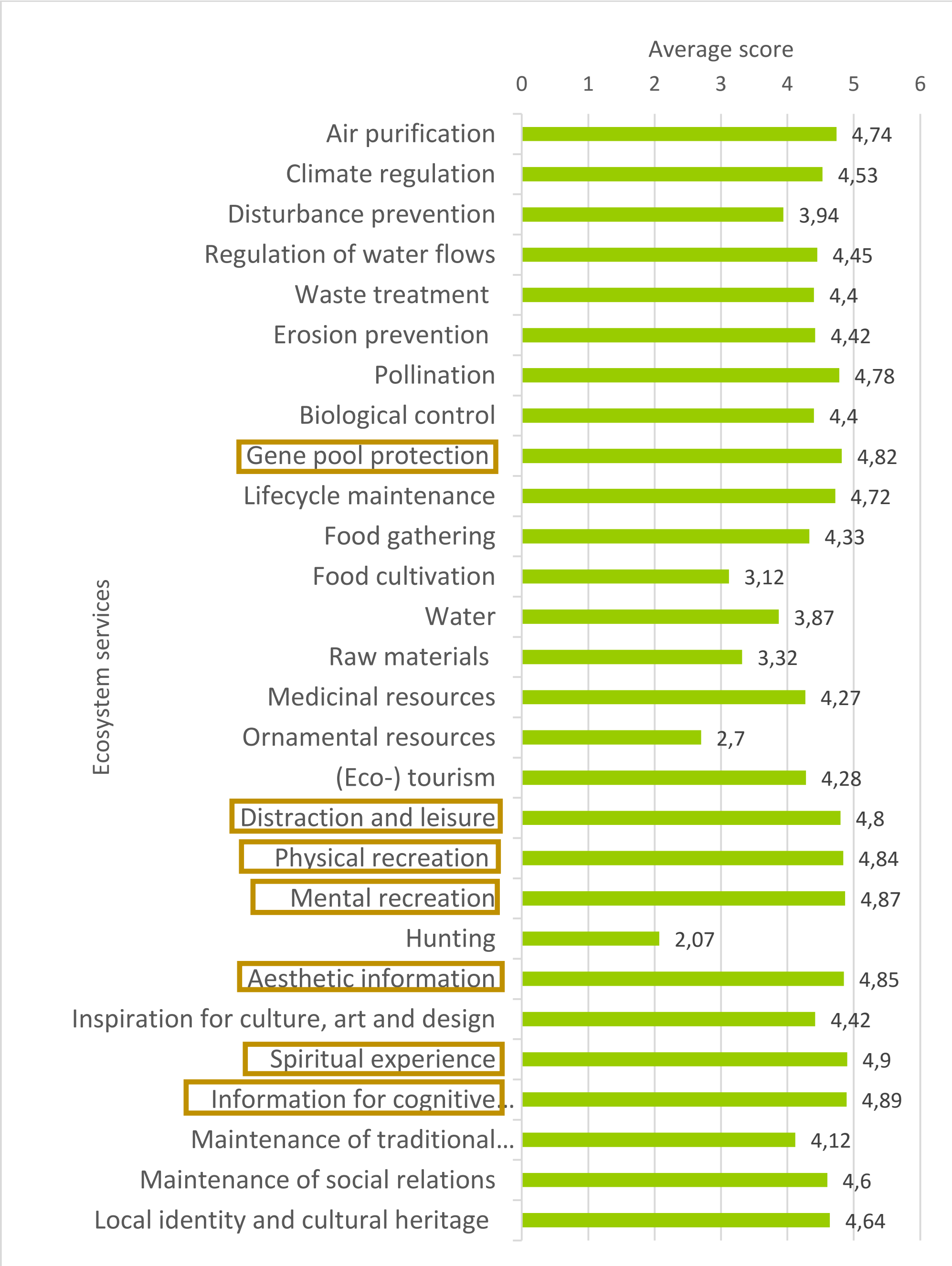

Figure 3.2 Average score of ecosystem services provided by the natural park of Sant Llorenç (highest values are indicated by boxes).

We compared the relative importance of the four categories of ecosystem services (Figure 3.3). Surveyed visitors gave on average the highest value to habitat services (4.77), followed by regulating services (4.46) and cultural ones (4.44). Provisioning services were the least valued with a relative importance of 3.60. As a kind of sensitivity analysis, we analysed the negative influence on the average score of the two least valued services in each category. In particular, we calculated the relative importance of the category cultural services without considering "hunting" which gave a score of 4.66, causing cultural services to be ranked as second most valued. The relative score of the category provisioning services without the "ornamental resources" was 3.78, which does not alter its ranking.

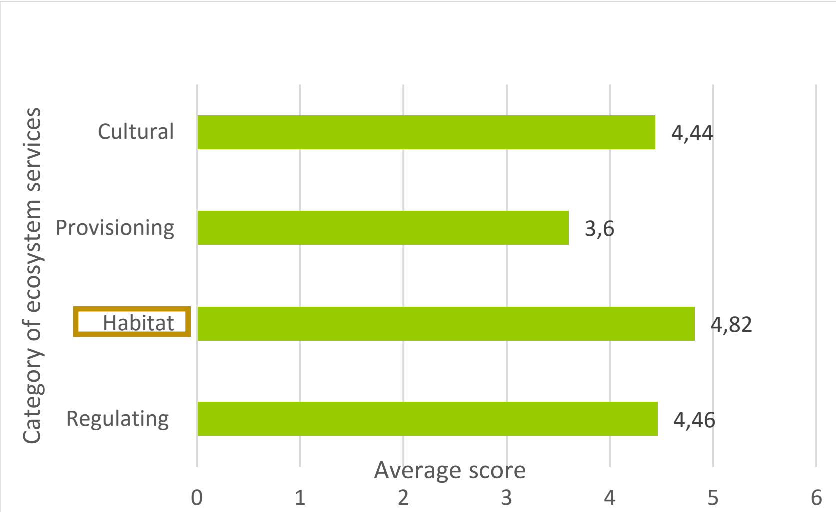

Figure 3.3 Average score of categories of ecosystem services provided by the natural park of Sant Llorenç (highest value indicated by a box).

3.3.4 Effects of visitor characteristics on the valuation of ecosystem services

Our survey sample (N=200) was composed of adult visitors, aged from 18 to 86 years (with mean = 40.4 years; standard deviation = 15.04), and with 71.5 were men and 28.5% women. This reflects, among others, that men use the park more than women, notably for hiking and cycling. Half of the visitors (50%) lived in municipalities that partly overlapped with the natural park. The majority (81%) of visitors lived in a city and 19% in a village. A little over half of respondents (58.5%) had a higher educational degree (superior professional or university formation). Few people had a job related to nature (only 14%). More than one third (34.5%) visited the natural park at least one time per week and 71.5% visited the natural park at least once a month. More than a half of surveyed people (53.5%) had at least once visited other natural parks apart from the three closest parks (Montserrat, Montseny and Collserola) at least once a year. One third (33.5%) were members of an environmental organization, organic food cooperative or hiking, climbing or biking group. More than a half of the visitors (62%) indicated that they consumed sometimes, frequently or always organic agriculture products.

Next we performed the multivariate regression analysis to test which factors explain variation in scores obtained by ecosystem services. The results in Table 3.3 show that significant explanatory variables in the average valuation of all ecosystem services (variable MeanES) are: *i)* Age, where older people on average valued ecosystem services more ($P \leq 0.05$); and *ii)* EduLevel, where people who had completed secondary education on average valued ecosystem services more ($P \leq 0.05$).

The regression analyses of the scores of specific ecosystem services indicate that certain visitor characteristics explain the valuation of individual ecosystem services. The most important ones are: *i)* people living in municipalities that had part of their area inside the park (Munipark) on average value more “gene pool protection” ($P \leq 0.1$), “medicinal resources” ($P \leq 0.1$), “eco-tourism” ($P \leq 0.1$), and “inspiration for culture, art and design” ($P \leq 0.05$); *ii)* people living in cities (Cityvillage) on average value more “disturbance prevention” ($P \leq 0.05$), “waste treatment” ($P \leq 0.05$), “erosion prevention” ($P \leq 0.1$) and “pollination” ($P \leq 0.05$); *iii)* people who were not members of environmental associations (EnvAssociation) on average value more the “regulation of water flows” ($P \leq 0.05$), “distraction and leisure” ($P \leq 0.1$), “mental recreation” ($P \leq 0.1$) and “aesthetic information” ($P \leq 0.05$); *iv)* people with a job unrelated to nature (Relnature) on average value more “food gathering” ($P \leq 0.1$) and “hunting” ($P \leq 0.05$); and *v)* older people (Age) living in villages (Cityvillage) on average value more “hunting” ($P \leq 0.1$).

In addition, we repeated the regression for a sub-sample constructed by taking the outliers. The results show that for the more polarized data with the outliers more factors are found to be statistically significant, which is what one would expect. In addition, we find for some ecosystem services similar factors (notably Age, Relnature, EduLevel and EnvAssociation, which can be seen as confirming robustness of the respective findings, while for others new factors appear. Generally, Relnature and to a lesser extent EduLevel appear as more important factors in the outlier analysis. Differences in valuing all ecosystem services (MeanES) can be explained largely by whether people work or not in a job related to nature (Relnature) and their age (Age). People with a job unrelated to nature and older people on average value ecosystem services more.

Table 3.3 Ordinary least square multiple regressions results

Service	Significant coefficients[1] (Complete sample)	Significant coefficients (Outlier analysis[2])
Air purification	Age**(+)	Age**(+)
Climate regulation	Age**(+)	Age**(+), **Relnature**(-)**
Disturbance prevention	Age*(+), **Income2****(+), **Cityvillage**(-)**, Ecoproducts2*(+)	**Relnature***(-)**
Regulation of water flows	Envassociation**(-)	-
Waste treatment (water purification)	Cityvillage**(-)	**Relnature**(-)**
Erosion prevention	Age***(+), Cityvillage*(-)	Age***(+), **EduLevel*****(+), **Relnature**(-)**
Pollination	Cityvillage**(-)	**Relnature**(-)**
Biological control	-	Age*(+), Munipark*(+), **Relnature**(-)**
Gene pool protection	Munipark*(+)	**Relnature***(-)**
Lifecycle maintenance	-	**Cityvillage***(+)
Food gathering	Age**(+), Relnature*(-)	Age***(+), **Cityvillage***(+), **Relnature***(-)**
Food cultivation	**EduLevel***(+)	-
Water	-	Age*(+)
Raw materials	-	**Relnature**(-)**
Medicinal resources	Age**(+), Munipark*(+)	Age*(+), **Relnature*****(-)
Ornamental resources	**Sex****(+), Age**(+), **EduLevel*****(+)	-
(Eco-) tourism	**EduLevel****(+), Munipark*(+)	**Relnature*(-)**
Distraction and leisure	EnvAssociation*(-)	**Relnature**(-),** Envassociation***(-)
Physical recreation (sport)	-	-
Mental recreation	Age***(+), Cityvillage*(-), EnvAssociation*(-)	Age*(+), Munipark*(+), EnvAssociation*(-)
Hunting	Age**(+), **Cityvillage***(+), **Relnature**(-)**	-
Aesthetic information	Age**(+), EduLevel**(+), EnvAssociation**(-)	Sex*(-), Age**(+), **EduLevel*****(+)
Inspiration for culture, art and design	Age***(+), Munipark**(+)	Age***(+), **Cityvillage****(+), **EnvAssociation**(-)**
Spiritual experience	Age***(+), EduLevel**(+)	Age**(+), **EduLevel****(+), Relnature**(-)
Information for cognitive development	-	-
Maintenance of traditional knowledge	Age*(+)	Age***(+), **EnvAssociation*(-)**
Maintenance of social relations	-	Age***(+), **Relnature***(-), EnvAssociation***(-)**
Local identity and cultural heritage	Age**(+), Income2**(+), EduLevel**(+)	Age**(+), **EduLevel*****(+)
MeanES (we include all)	Age**(+), EduLevel**(+)	Age **(+), **Relnature** (-)**

Notes: * Significant at ≤10%; ** Significant at ≤ 5%; *** Significant at ≤ 1%

[1] Variables in bold indicate that the coefficient>0.5. The coefficient sign is indicated in brackets.

[2] This is based on a sub-sample with the lowest and highest values of the mean ES. This includes 53 people (±25%); 25 with a mean ES value below 3.75 and 28 with this value above 4.80.

3.3.5 Conflicting views or preferences between stakeholders

Several conflicting views between stakeholders were identified. The main conflicting view mentioned by 19 stakeholders (76% of interviewees) relates to an actual historical conflict between conservationists (linked especially with hiking groups) and park managers on the suitable park management approach. Conservationists have persistently argued that the natural park of Sant Llorenç should prioritize nature conservation and limit public use of it, while natural park managers have tended to enhance touristic and cultural activities inside the park. This conflict had materialized in many occasions during the more than thirty years of existence of the natural park. For instance, the establishment of forests tracks for fire extinction or the road to connect the villages Mura and Monistrol de Calders. In both cases, conservationists argued that these actions promoted easy access of people to previously isolated natural areas. Other conflicting views were: *i)* between hunters and visitors as stated by 10 stakeholders (40%): the latter considered that hunting was not a proper activity to be developed in a natural park; *ii)* between owners and visitors because the latter entered inside the crop fields of the first and damaged these (n=9, 36%); *iii)* between motorcyclists and visitors because the former did not comply with the legal restrictions and used prohibited paths.

3.4 Interpretation of the results and comparison with other studies

3.4.1 People's awareness of benefits provided by the natural park

3.4.1.1 Stakeholders' perceptions of ecosystem services

Our results suggest that stakeholders are aware of a wide range of benefits provided by the natural park to human well-being. Cultural and habitat services were the most perceived ecosystem services, while regulating services were the least perceived. These results are in consonance with other studies that found that cultural services are the most perceived for respondents in green urban areas of Spain (Casado-Arzuaga et al., 2013) and that regulating services are sometimes overlooked (Agbenyega et al., 2009). López-Santiago et al. (2014) also found a high perception of cultural services in Mediterranean areas of Spain, although regulating services were also high perceived. In our case, the most perceived ecosystem services were "aesthetic information" and "gene pool protection", something which does not come entirely as a surprise considering that these are perhaps the most basic reasons to protect a natural area.

Another not so surprising result is the frequent mention of the service "local identity and cultural heritage". One reason for this is that local people feel this attachment through identity. Another is suggested by our ethnographic research through non-participant observation, namely the long tradition of hiking groups from surrounding cities. A third reason is related to the first two, namely the presence of rough mountains (including the emblematic la Mola peak) near a densely populated urban region. “Local identity and cultural heritage” was also identified to be a main ecosystem service in the case of home gardens in Catalan Pyrenees (Calvet-Mir et al., 2012) and in the case of maintaining “drove roads” in the Serranía de Cuenca of Spain (López-Santiago et al., 2014).

The lack of knowledge about certain regulating services may be due to the lack of understanding of how the underlying ecological processes operate. For instance, people are aware of biodiversity conservation, i.e. “gene pool protection”, but the ecological processes that lead this biodiversity to control pests, i.e. “biological control”, are complex and not widely perceived.

These results reflect the changes in the relation between humans and the environment in rural areas, particularly near cities. Some urban dwellers perceive these areas as associated with the traditional identity of their families, and others mainly as places for relaxation and recreation, less than as providers of ecosystem services like regulating and provisioning that indirectly support their wellbeing. This is possibly linked to the fact that the use of these landscapes is being increasingly devoted to uses within the tertiary sector instead of the traditional farming or forest management practices.

3.4.1.2 Visitors’ valuation of ecosystem services

The category of habitat services was the most valued by visitors, followed by cultural and regulating ones, while provisioning services were the least valued. We think that the most valued services respond to the characteristics of the urban dominant profile of visitor that comes to the natural park to spend leisure time and disconnect from the urban lifestyle, except for “gene pool protection” and “information for cognitive development”. In the case of the former, we think that the instrument of protection fosters the importance people attribute to this service. In the case of the latter, it is possible that respondents are aware that research on nature and education about nature can contribute to better knowledge about the natural park. Thus, the general ecosystem service “information for cognitive development” reinforces the perceived importance of the service “gene pool protection”.

The importance attributed to cultural services and “gene pool protection” by urban visitors is also obtained in other studies such as Martín-López et al. (2012) in green areas of Spain. Urban visitors valued provisioning services little, which can be explained by the fact that the food or water they consume or the natural resources they use are not obtained from the natural park. Moreover, urban visitors do not develop rural traditional activities like agriculture, logging or gathering medicinal plants. This low importance attributed to provisioning services illustrates the general decoupling of urban populations from ecosystems in terms of a material dependence. The provisioning service most valued was “food gathering” of mushrooms and wild fruits. This reflects a concrete activity undertaken by visitors that is linked to provisioning services.

As we have mentioned, “gene pool protection” was highly valued, like in other studies (Agbenyega et al., 2009; Lamarque et al., 2011; Martín-López et al., 2012). The process of local people being involved in civil and political movements aimed at protecting the area may partly explain why they highly value biodiversity conservation. The importance of biodiversity conservation is also reflected in the highest value assigned to “pollination” services. Users see the natural park as a place that contributes to the conservation of the recession of species (like bees) that provides this ecosystem service. Surprisingly perhaps, this service was the highest valued regulating service by visitors even though it was not mentioned by stakeholders. We conclude from this that when people do not by themselves identify certain ecosystem services this does not necessarily mean that they would judge them as unimportant if asked so. Such effects are also reported by Lamarque et al. (2011) and Casado-Arzuaga et al. (2013).

3.4.1.3 Lack of knowledge and misunderstanding of the term ecosystem services

Despite the fact that many respondents perceive the presence and importance of ecosystem services, we found that the term “ecosystem service” was unknown to most stakeholders and visitors. The concept was not unanimously accepted and its use often led to confusion because people commonly related the word “service” to human activities. This type of result was also found in Plieninger et al. (2013) and Lamarque et al. (2011). In the latter study some respondents regarded the term “ecosystem services” as too anthropocentric and biased, insufficiently indicating that rather than nature providing services to humans the latter used the first. We therefore recommend using the term “ecosystem service” only or mainly with experts while talking about “benefits that nature provides to humans” when communicating with laypersons.

3.4.2 Explanatory power of visitors’ characteristics in socio-cultural valuation

The regression analyses show that the main characteristics of visitors that influence socio-cultural valuation are: *i) age:* older people on average value more ecosystem services; *ii) place of residence:* people living in municipalities with part of their area inside the protected area on average value more “gene pool protection”; *iii) education:* people with higher scholar education on average value more ecosystem services, and *iv) place of residence:* people living in cities on average value more certain regulating services.

Regarding the first finding, older people have witnessed the emergence of protection of the natural park. As we have shown, the creation of the natural park was not a decision planned by politicians but rather the result of historical requests of local people to protect the area (Aguilar, 2012). Consequently, such older people may value more the contribution of the natural park to their well-being. Local requests in the past to protect the area seem to be consistent with the high valuation of “gene pool protection” by people living in municipalities with part of its area inside the protected area. This result seems surprising if we compare it with other studies finding that protected areas can evoke strong resistance among local population against protection measures, because they are seen as limiting their activities (Stoll-Kleemann et al., 2001). In fact, such resistance also occurred in the early years of protection of the natural park (Aguilar, 2012). However, over time local inhabitants have tended to accept the restrictions for protection. Moreover, many of the traditional land uses have been abandoned and the local economy has become more dependent on the tertiary sector, so that it has in fact become less restricted by nature protection.

Our results show that people with higher education value more ecosystem services. This is likely due to education increasing the capacity of people to understand and acknowledge the capacity of ecosystems to provide services, as is also found in Martín-López et al. (2012).

People living in cities value higher certain regulating services. Similar results were found by Martín-López et al. (2012) for various Spanish regions and by Casado-Arzuaga et al. (2013) for the Bilbao Metropolitan Greenbelt (Spain). We think that people are aware of the negative impacts on nature caused by the Barcelona Metropolitan Region, such as nature fragmentation and pollution, and that many regard the ecosystem of Sant Llorenç as somehow compensating for such impacts.

3.4.3 Are natural parks near urban areas just landscapes for décor and leisure?

Here we want to examine the social perception of the natural park of Sant Llorenç considering the elicited preferences of stakeholders and visitors regarding ecosystem services. Buijs et al. (2006) note that three images of landscapes can be identified: 1) the Arcadian image, which is the rural idyll characterized by harmony between human and nature; 2) the wilderness image, which is the non-regulated and autonomous appearance of nature that is not subject to human influence or control; and 3) the functional image, which is a landscape that serves primarily life-support and utilitarian services.

The fact that the ecosystem services most valued are “spiritual experience”, “information for cognitive development”, “mental recreation”, “aesthetic information”, “physical recreation”, “gene pool protection” and “distraction and leisure”, suggests that the second, wilderness image is dominant among the perception of the natural park by visitors. Here landscape may be understood as a décor and leisure space with little to no need for human intervention.

In addition, regulating services were highly valued while a low value was attributed to provisioning services. These findings are in accordance with the trends over the past century of abandonment of traditional rural activities (Plieninger et al., 2013) which imply moving away from a functional (the third) image of the natural park. As opposed to the low average valuation of provisioning services by visitors, many stakeholders pointed out the necessity to enhance certain provisioning services, notably “food cultivation” and “raw materials”. They claimed to enhance “food cultivation” through organic agriculture due to its multifunctionality in increasing biodiversity, the maintenance of traditional knowledge, prevention of fires, aesthetic values and production of an alternative local food network which emphasizes local scale and proximity. Such high valuation of multifunctionality of agriculture is also found in Casado-Arzuaga et al. (2013), Calvet-Mir et al. (2012) and Lamarque et al. (2012). Stakeholders stressed the value of enhancing the provision service of “raw materials” like wood in order to diminish the risk of forest fires, to generate more open areas that contribute to enhance biodiversity and as an economic activity in itself.

Next, the service “maintenance of traditional knowledge” was also lowly valued by visitors but emphasized by stakeholders as an ecosystem service that should be enhanced through the recuperation of rural traditional activities. This recuperation permits the maintenance of local ecological knowledge systems that have considerable value for sustainable ecosystem management and biodiversity conservation in industrialized countries (Gómez-Baggethun et al., 2010b, 2010c; Otero et al. 2013).

Boada and Otero (2006) reported a negative public perception of activities developed by local peasants, shepherds, and foresters in protected Catalan ecosystems. However, our study shows that stakeholders understand the role of human intervention (e.g. through agriculture or logging) in maintaining the current cultural landscape. This may be due to stakeholders’ awareness of problems associated with land abandonment, such as an increasing risk of fires due to the accumulation of biomass (Terradas, 1999; Rudel et al., 2005; Lasanta et al., 2006) or the loss of biodiversity (Atauri and de Lucio, 2001). Thus, stakeholders requests for an increase in rural activities can be interpreted as understanding landscape as the Arcadian image. Our results are in line with studies in France and the Netherlands that suggest that a shift from a functional image of nature and landscape to a more Arcadian and wilderness one has taken place (Buijs et al., 2006).

3.4.4 Conflicting views among stakeholders due to diverging socio-cultural preferences

Our findings indicate two main conflicting views, one related to tourism and the other to hunting. Contrarily to what have been found in Martín-López et al. (2012) and Casado-Arzuaga et al. (2013), where “tourism” was highly valued, in our socio-cultural valuation it was not much valued. Nevertheless, many stakeholders identified tourism as an ecosystem service that should be better managed; notably, the number of visitors coming to the natural park should be controlled and even reduced. Many stakeholders thought that tourism should be redirected to surrounding villages that could benefit economically from it, also since its inhabitants valued “eco-tourism” relatively much. The Statutory Board of the natural park has promoted policies enhancing cultural and touristic activities that increase the number of visitors considerably. These policies are criticized by conservationists (environmental associations and hiking groups) who argue that a natural park should prioritize nature conservation and limit public use. This illustrates clearly conflict viewpoints regarding which uses should be promoted in the natural park. There is a need to develop an urban green belt around the natural park that is capable of absorbing the large amount of urban people that uses the park for leisure purposes.

Hunting represents another conflicting view reported by stakeholders. This is reflected by the low value given by visitors to this ecosystem service. They indicated that this activity was not appropriate inside a natural park as it negatively affects quietness required for biodiversity conservation and leisure. Especially older people in villages value “hunting”, likely as they are the ones who undertake this activity. Despite the low average score obtained in the socio-cultural valuation, some stakeholders argued that hunting could be useful in controlling the size of the population of wild boars and minimize negative impacts these provoke on crops. Our results show a trade-off between “hunting” and cultural services related with leisure activities that should be taken into account by decision-makers when considering the different types of benefits and users of the natural park.

3.5 Conclusions

The present study has shown the importance of assessing social perception of and values attributed to protected areas in order to improve their design and management. Conducting a socio-cultural valuation of ecosystem services reveals how important people think the natural park is for improving their well-being. This holds true regardless of whether one deals with natural parks or other types of management regimes of nature areas. We have presented an original approach consisting of four methods that provide complementary information about the issue studied. Application of this approach to the natural park case in Catalonia, Spain has demonstrated the following: *i)* the term ecosystem service is unknown by the general public and frequently misunderstood because it is related to human actions to improve nature; *ii)* identifying ecosystem services by consulting stakeholders reveals people’s awareness of a wide range of benefits provided by the natural park; *iii)* habitat and cultural services are the most valued, especially those related to leisure activities; stakeholders are aware, though, of the importance of traditional rural activities (linked to provisioning services) in maintaining the Mediterranean cultural landscape and also as a tool to diminish the risk of forest fires, to increase biodiversity and maintain traditional knowledge; *iv)* age, education and place of residence are the main characteristics of respondents that affect their socio-cultural valuation of ecosystem services; *v)* conflicting viewpoints among stakeholders can be identified from diverging socio-cultural preferences.

Our approach consisted of four methods, namely literature review, non-participant observation, semi-structured interviews and valuation survey. These are complementary and served the following particular functions: the first three methods were used in a preliminary phase to provide the basis for implementing the fourth method; methods 3 and 4 provided complementary information to generate insights about perceptions and values associated with ecosystem services; finally, methods 2 and 3 jointly identified potentially conflicting viewpoints on park management. We think that this set of methods can equally be applied to other cases to generate useful insights for park management or other instruments of protected areas. The reason is that they can tackle the complexity of such areas well, in terms of multiple ecosystem services, multiple stakeholders and multiple levels of governance.

The way people perceive and value protected landscapes clearly depends on various factors, including functional ties with the landscape, individual experiences and situations, where and how people live, (changing) land uses, the knowledge about the historical process that led to the protection of the area, and the diffusion of knowledge in society about the problems affecting the natural park. In view of this multidimensionality of valuation, it is important to complement socio-cultural valuation assessments with other methods, such as interviews and non-participant observation, as these provide complementary information on which ecosystem services need to be enhanced by park management. One possible shortcoming of the socio-cultural valuation approach adopted here is that rather high values were obtained for a large number of respondents and for many ecosystem services. This may reflect their preferences, but it may also be the case that the combination of positive formulation of ecosystems services and Likert scale enhance high values. As part of future research, one could examine other ways to elicit preferences, such as a so-called “Pebble Distribution Method”, which involves giving a limited number of points to participants which need to be divided between all items, in our case ecosystem services (Sheil et al., 2002).

Appendix Table A.3.1 Categorical and continuous variables of the multivariate regression analysis and descriptive statistics

Categorical variables			
Variable	**Definition**	**Values**	**Results (%)**
Sex*	Dummy variable: Sex of the respondent	0= Woman	28.5
		1= Man	71.5
Income2*	Dummy variable: whether the respondent has an income higher than 1000 euros per month	0= No	11.3
		1= Yes	88.7
Munipark*	Dummy variable: the respondent lives in a municipality with part of it inside the park area	0 = Outside park area	50
		1= Inside park area	50
Cityvillage*	Dummy variable: the respondent lives in a municipality with more than 20000 habitants[1]	0 (yes)= City	81
		1 (no)= Village	19
Education	Discrete variable: respondent's level of education	0= No studies	0
		1= Primary	9
		2= Secondary	11.5
		3= Bachelor/Medium professional formation	21
		4= Superior professional formation	15
		5= University	43.5
EduLevel*	Dummy variable: respondent's level of education	0= No studies and primary	9
		1= Secondary, Bachelor/Medium professional, Superior professional formation and University	91
Relnature*	Dummy variable: Whether the respondent has a job related with nature	0= No job related with nature	86
		1= Job related with nature	14
VisitSant	Discrete variable: respondent's frequency of visits to Sant Llorenç del Munt	1= One time or more per week	34.5
		2= Every two weeks	9
		3= One time per month	28
		4= One time per year	14
		5= Unique visit	14
Visitothers	Dummy variable: whether the respondent's visits other natural parks	0= No visit other parks	46.5
		1= Visit other parks	53.5
Association	Discrete variable: the type of association where the respondent belongs to	0= No association	66.5
		1= Environmental organization	4.5
		2= Hiking, climbing, biking group	26
		3= Cooperative	1.5
		4= Others	1.5
EnvAssociation*	Dummy variable: The respondent is a member of an association	0= No member	66.5
		1= Member	33.5
Ecoproducts2*	Dummy variable: the respondent consumes ecological agriculture products	0= No consumption	38
		1= Consumption	62

Continuous variables				
		Mean	**Min**	**Max**
Age*	Respondent's age in years	40.41	17 (18)	86
IncomeRealHH*	Respondent's monthly income in his/her household divided by the number of people living in the household	1047.55	0	4000

Notes: * Variables used in the multivariate regression analysis.

Chapter 4. Participation in protected areas: a social network case study[11]

4.1 Introduction

Local participation in governance of protected areas is considered to be important to natural resource management and biodiversity conservation (Dudley, 2008; Borrini-Feyerabend et al., 2013). Participation has been defined by Wesselink et al. (2011) as any type of inclusion of non-state actors, both members of the public or organized stakeholders, in any stage of governmental policy making. Several studies have emphasized the need for participation in governmental decisions (Fiorino, 1990; Blackstock and Richards, 2007; Wesselink et al., 2011; Fischer, 1993; Reed, 2008). Various reasons for these have been identified: participation assures more legitimate decisions, thus enhancing public credibility in governments; it reduces potential conflicts between different stakeholders; it increases the variety of information that contributes to better decisions; and it counters the power of incumbent interests by allowing all those affected by a decision to influence the associated decision process.

Before the 1980s, communities tended to be excluded from public decision-making, or their participation was even regarded as counterproductive to natural resource management (Ruíz-Mallén et al., 2013). This approach was challenged by studies that stressed the inclusion of local people in natural resource governance (Hutton et al., 2005). The rights and need for local participation in decision making into protected areas was articulated at successive world congresses on National Parks and Protected Areas, particularly the third in 1982 and the fourth in 1992 (McNeely, 1992), as well as in the Convention of Biological Diversity (CBD, 1992). Recently, active stakeholder participation has been recognized as a key factor of effective area protection in The Programme on Work on Protected Areas (PoWPA) of the CBD (Dudley, 2008) and in the 2020 Biodiversity Strategy (European Union, 2011).

Participatory initiatives for natural resource management nowadays include stakeholder analysis, that is, the process of identifying individuals or groups that are likely to affect or be affected by conservation efforts (Freeman, 1984; Reed et al., 2009). This type of analysis has responded to the failure of many past conservation plans caused by paying insufficient attention to the interests and characteristics of

[11] This chapter also appears as: Calvet-Mir, L., Maestre-Andrés, S., Molina, J.L., van den Bergh, J.C.J.M. 2015. Participation in protected areas: a social network case study in Catalonia, Spain. *Ecology and Society* 20(4):45 http://dx.doi.org/10.5751/ES-07989-200445.

stakeholders (Grimble and Wellard, 1997). There is now increasing recognition and understanding of how stakeholders can or should influence natural resource management (Burroughs, 1999; Prell et al., 2009). However, stakeholder analysis has two main limitations. First, stakeholders are usually identified and categorized through a subjective assessment of their relative power, influence and legitimacy leading to a misrepresentation of stakeholders (Frooman, 1999). Second, methods for stakeholder analysis often overlook the role communication networks can play in categorizing and understanding stakeholder relationships (Prell et al., 2009). Social network analysis is a tool that can help to overcome these shortcomings by providing insights into the social structure of stakeholders (Prell et al., 2008).

We studied the social network of communication with regard to the natural park of Sant Llorenç del Munt. We based our research on previous studies suggesting that the exchange of knowledge and information is crucial for effective governance of natural resources (Bodin and Crona, 2009) and that social network analysis may disclose the communication networks of stakeholders (Prell et al., 2011). Social networks are observable social structures (Bodin et al., 2006) made up of individuals or organizations tied by one or more specific types of interdependency, such as common interests or communication exchange. Social network analysis may demonstrate the existence and importance of social drivers supporting natural resource management (Beilin et al., 2013), reveal structural characteristics of networks that articulate the patterns of connectivity between actors, and influence natural resource management outcomes (Bodin and Crona, 2009).

In this study, we combined quantitative and qualitative data collection to undertake social network analysis, with the aim to map local exchange network of information and examine to what extent this exchange of information is being reflected in formal participatory bodies. The concrete objectives of the research are: 1) to assess the structure of the communication network associated with the natural park, 2) to compare the informal communication network between stakeholders with the formal participatory bodies of the natural park, to see whether the latter represents the actual social network; and 3) to select participants for later analyses aimed at assessing the adequate governance structure of the natural park.

4.2 Study site and methods

4.2.1 Description of the governance of the natural park

The natural park is managed by the Diputació de Barcelona, a regional administration corresponding to the territorial area of the Barcelona Province. The governance of the natural park includes two participatory bodies: the Coordinating Council and the Advisory Committee. The Coordinating Council is the formal institution that guaranties the participation and collaboration in park management of the different public administrations with competencies in the specific areas. It meets every six months and is composed of representatives of the Diputació de Barcelona, representatives from the council of each municipality that has part of its area inside the protected area, a representative of the Catalan Government (Generalitat de Catalunya), and a representative of the park's Advisory Committee. The Diputació de Barcelona proposes most of the actions and plans to be developed whereas the other public administrations have to validate these proposals, but are also allowed to propose initiatives. The Advisory Committee is an informative public meeting of stakeholders and park managers held every six months. Its main objectives are twofold: 1) Inform stakeholders about the policies and actions implemented or planned to be executed; and 2) collect the comments of stakeholders on the issues presented, even though these comments are not binding. The Advisory Committee was established in 1986 in order to "guarantee stakeholder participation, understanding this participation as a non-professionalized and unpaid voluntary action, and aimed at facilitating the suitability of decision-making to social demands" (Diputació de Barcelona, 1997). It is composed by representatives of the Diputació de Barcelona, the Coordinating Council, and the various social, economic, scientific, cultural and conservationist organizations with a stake in the management of the natural park. Our research focuses on analyzing the Advisory Committee as a formal participatory body, given that every single stakeholder can join, and compare it with the existing informal network of communication among stakeholders.

4.2.2 Methods of data collection

We collected data on the natural park of Sant Llorenç del Munt (Figure 1.1) between January and November 2013 and between April and September 2014. Data collection included non-participant observation, review of documents, semi-structured interviews and on-line/telephone surveys.

4.2.2.1 Non-participant observation

We used non-participant observation techniques to establish contact with the community, local culture and local social organization in a non-active way (Bessette, 2004). We undertook several trips from January to June 2013 in order to visit the twelve municipalities that have part of their area within the natural park, and we had informal talks with individuals or groups in order to know the relationship of each municipality with the natural park. In addition, we stayed two weeks in July and August 2013 in a farmhouse inside the park. This stay gave us knowledge about the situation of people living inside the protected area and their perceived role of participation in natural park governance. We also participated in a meeting of the Advisory Committee to observe how this participatory body works and how stakeholders interact.

4.2.2.2 Review of documents

We reviewed all the available documents about attendance to meetings of the Advisory Committee from 2008 to 2014 (in total 11 sets of minutes) to define the stakeholders within the natural park and the categories they belong to. We further listed the number of times each stakeholder attended these meetings.

4.2.2.3 Semi-structured interviews

We conducted semi-structured interviews (N=25) concerning the participation mechanisms in the natural park with relevant stakeholders. We also asked them to name the people they considered important for natural park management. We selected stakeholders who had an interest in natural park management because they affect or are affected by decisions regarding it (Reed et al., 2009). They were selected on the basis of reputation and recommendation from a small pool of initial stakeholders following a snowball sampling strategy. This uses a small pool of initial informants to nominate, through their social networks, other participants who meet the eligibility criteria and could potentially contribute to a specific study (Newing, 2011). To avoid selection bias, we interviewed people from all the sectors present in the natural park, e.g. mayors, park managers, farmers, forest owners, etc. All the interviews were recorded with previous consent.

4.2.2.4 Survey

With 65 stakeholders from the natural park, we conducted an on-line/telephone questionnaire to assess how stakeholders are connected and communicating among themselves. We selected the sample based on: 1) non-participant observation, 2) people attending at least three Advisory Committee meetings (based on reviewed documents), 3) interviewed stakeholders, and 4) stakeholders considered important for natural park management by the interviewees. We obtained a list of 117 people that were supposed to compose the social network of the natural park. In order to add relevant people or delete people that were no longer linked to the natural park we sent the list of stakeholders to all the participants (n= 28) of the Advisory Committee meeting held in November 2013 and five key informants selected by ourselves from interviews. They could provide comments on the basis of which we came to a final selection of people connected to the natural park. Finally, ten people checked the list and we ended up with a final list of 105 people that was reduced to 98 due to inaccessibility to personal contact details, i.e. e-mail or telephone number. We further established 12 different categories of stakeholders corresponding to sectors present in the natural park such as local administrations (mayor and councilors of environment), park managers, park employees, representatives of conservationist, civic and leisure organizations, workers in the agricultural, scientific, tourism, environmental education and forestry sectors; and other companies related to the natural park. In the survey, we specifically asked people "With whom do you communicate about issues related to policies and natural resource management in the natural park of Sant Llorenç del Munt?", and "With whom do you have any conflict?" After all names were listed we asked information about sex and stakeholder category where the person listed pertained. We also asked stakeholders to introduce their personal data (name, stakeholder category, and sex). Respondents were informed that their responses would be anonymized because of the sensitivity of the question on conflicts, thereby trying to mitigate the reliability of responses (Marsden, 1990). From the 98 people approached, 65 responded (a response rate of 66.32%).

4.2.3 Methods of data analysis

As part of the social network analysis, we used information from the survey to 1) explore the network of communication of Sant Llorenç del Munt, 2) calculate two individual centrality network measures (*indegree* and *betweenness*), and 3) make clusters of actors that have the same ties to and from the same actors in the network. We assessed the network of communication of Sant Llorenç del Munt using the survey question "With whom do you communicate about issues related to policies and natural resource management in the natural park of Sant Llorenç del Munt?" Data was handled using the software UCInet6-Netdraw for Windows (Borgatti et al., 2010).

We calculated four network-level measures. These measures are informative about the general features of the network, paying attention at the same time to the level of cohesion/fragmentation and the existence of eventual leaders in terms of connections (Borgatti et al., 2010):

1) *Size*, or number of actors in the network.
2) *Number of components*, or the number of connected subgraphs in which all actors are directly or indirectly in contact with each other.
3) *Density*, or the number of links in the network, expressed as a proportion (from 0 to 1) of the maximum possible number of links.
4) *Indegree network centralization index*, or the tendency for a few actors in the network to receive many links or nominations (expressed in percentage).

We also calculated two individual-level centrality measures, both of them widely acknowledged by the literature as reliable indicators of both prestige (*indegree*, Wasserman and Faust, 1994), and brokering capabilities (*betweenness*, Burt, 2003).

1) *Indegree*, or the number of nominations that a person receives on other people's lists. For example, if four people mentioned one informant when asked to list the name of who he/she communicated about policy and natural resource management issues in natural park of Sant Llorenç del Munt, then the informant would have an indegree of four. It is a measure that represents more popular/well-connected stakeholders in the network. We used *indegree* instead of *degree* (i.e. the number of links a stakeholder has using data as

symmetric) because literature point out that *indegree* is a more robust measure for assessing informal organograms and under conditions of missing data (Costenbader and Valente, 2003).

2) *Betweenness*, or how many times an actor rests on a short path connecting two others who are themselves disconnected. This indicates which stakeholders brokered across different stakeholder categories and disconnected segments of the network.

In addition we measured the level of *dyadic reciprocit*y, i.e., the extent of mutual nominations among stakeholders. We carried out a *core-periphery analysis* (Borgatti and Everett, 2000) which has been reported as a typical feature of social networks in general (Mcpherson et al., 2001), and useful for understanding performance in groups (Cummings and Cross, 2003). This measure served us to identify which actors belonged to the core and which belonged to the periphery of the network and to verify the relevance of the stakeholders interviewed, thus assuring representation from stakeholders belonging both to the core (n=13) and the periphery (n=12) of the network.

Next, we ran a *single-link hierarchical clustering analysis* to assess stakeholders' structural positions (Prell et al., 2008, Prell, 2011). This tool groups actors that have the same ties to and from the same actors in the network and thus can be considered to be more or less redundant within the network (Wasserman and Faust, 1994). As there is not a guideline in the number of clusters an analyst can obtain, we split the network in four meaningful clusters based on our previous knowledge of the site.

Finally, the question on "With whom do you have any conflict?" served us to generate a social network of conflict within the park in order to avoid selecting stakeholders who had conflicts in the past to join future participatory processes.

As part of the statistical analysis, we ran Spearman correlations to examine the association between the person's centrality in the communication network and the number of times s/he participated in meetings of the Advisory Committee between 2008-2014. To test the robustness of the analysis, we undertook a Wilcoxon rank-sum test using both measures of centrality and a binary variable, *AC*. This was coded as 1 if the person participated any time in a meeting of the Advisory Committee and 0 otherwise. Finally, in order to examine if there were any specific category of stakeholders that hold more centrality we looked at the mean indegree and betweenness of each category and based on these descriptive statistics we ran other Wilcoxon rank-sum test using both measures of centrality and a binary variable named *Park Employees.* This was coded as 1 if the person was an employee of the natural park and 0 otherwise. For the statistical analysis we used STATA 12 for Windows.

4.3 Results and discussion

4.3.1 The communication network of the natural park

There exists a network of communication composed of 238 stakeholders and structured in one single component (Figure 4.1). The network has a very low density (0.008) indicating that there are few ties between stakeholders. It has an indegree centralization index of 11.50%. This is low compared to that of a pure star network with a centralization index of 100%, indicating that the indegree of concentration in the distribution of indegree centralities among the actors is fairly low. This low index shows that the network does not have very central (dominant) stakeholders.

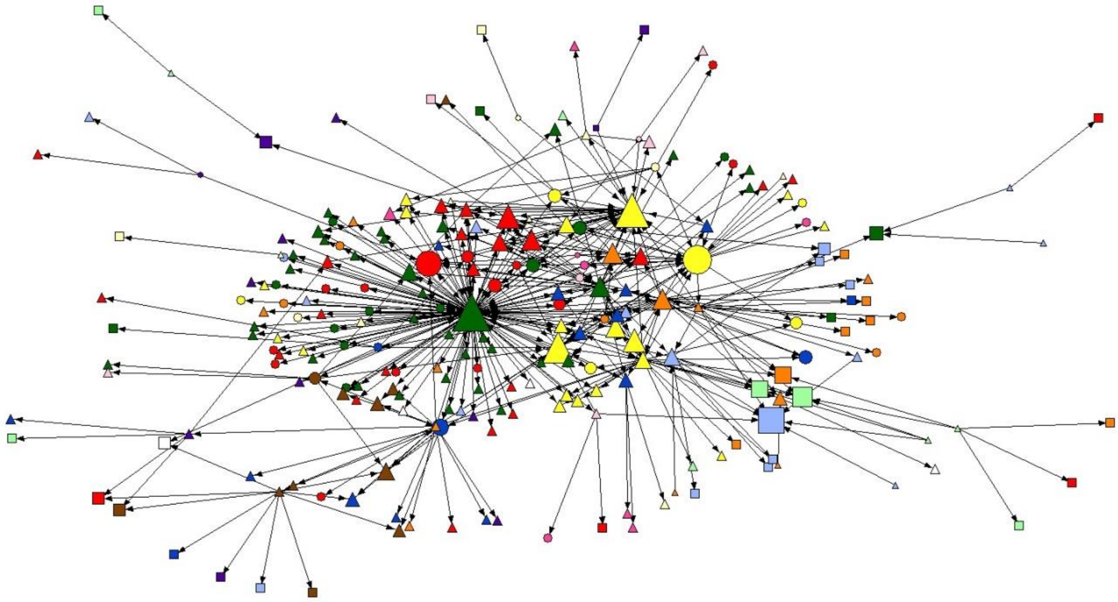

Figure 4.1 Communication network of the natural park of Sant Llorenç del Munt

Notes: The size of the nodes indicates the indegree, the shape the sex (circle for women, a triangle refers to men and a square institutions). In addition, the following colors are used to denote the various categories: Red (Local administration); Green (Park managers); Yellow (Park employees); Blue (Agricultural sector); Orange (Scientific sector); Purple (Civic sector); Pale green (Conservationist sector); Pale blue (Leisure sector); Pale yellow (Environmental education-Tourist sector); Brown (Forestry sector); Pink (Accommodation and restaurants); Pale pink (Other enterprises); White (Others).

The measures discussed indicate that the network of communication of the natural park of Sant Llorenç del Munt is not fragmented because there is only one component, but it is fragile because it is characterized by a low density. A main reason is that some people in the network are connected to only one other person. This is confirmed by the fact that dyadic reciprocity, i.e. the proportion of mutual nominations, is less than 9% (0,0829), indicating a low level of communication among stakeholders. Connection is very important because it guarantees access of information by many individuals and the building of relations of trust between people (Borgatti and Foster, 2003, Rishi, 2007). Our results from semi-structured interviews pointed out that participatory processes are not working properly because some agreed-upon decisions were finally not implemented. This generated a lack of trust in participatory processes and park managers, which likely contributed to a decrease of communication among certain stakeholders who usually conversed within the space of these processes. For example, one stakeholder said *"Participatory processes are not useful, nothing is implemented"* and other stated *"They are not operative […] finally I took distance from park management and participatory processes"*. This perceived lack of trust between some stakeholders could also increase the reluctance of collaborating with others (Ostrom, 1990, 2010). Moreover it could undermine the positive role that network connections could have in establishing reciprocity (Adger, 2003) or in increasing social memory (Bodin et al., 2006). With regard to the low degree of centralization of the park's social network, literature has characterized both advantages and disadvantages. In the case of the former, it may increase the exchange of different types of knowledge within the network engaging people into a continuous learning process whereby management of the natural park can be updated and adapted (Holling, 1978; Bodin et al., 2006; Prell et al., 2007). Our results showed a high diversity of stakeholders and organizations within the natural park, which may sow the necessary conditions for processes of cooperation and learning in decision-making. As several interviewees highlighted, the existence of multiple stakeholders involved in the natural park permits the integration of different perspectives necessary for a comprehensive management of the protected area. However in current conditions of mistrust, learning and adaptive capacities of the network might be weakened (Bodin et al., 2006). In terms of disadvantages, a low degree of centralization can hamper adaptive capacity to changing conditions because it may diminish coordination ability to cope with problems (Leavitt, 1951; Prell et al.; 2007). Nevertheless, some actors, notably the park director and several park employees, hold a high indegree (Figure 4.1) and also are the ones who hold major responsibilities for park management. This could overcome the lack of coordination assumed in a low centralized network.

The results suggest that an informal network of communication exists and holds a potential to deal with the management of the natural park. However, it is probably less effective than it could because of a lack of trust in the effectiveness of participatory bodies, such as the Advisory Committee, by some stakeholders. Several studies indicate that lack of trust is a major reason for ineffective natural resource management. This has been illustrated for such different issues as weed management (Graham, 2014), farm management (Hernández-Jover et al.. 2012), wild animal management (Davies and White, 2012), and management of marine protected areas (Ho et al., 2014). A common thread in these studies is the necessity to generate or rebuild trust between stakeholders and formal organizations.

4.3.2 Do participatory bodies represent the social network of the natural park?

Results from the Spearman correlations (Table 4.1) indicate that stakeholders who hold more indegree and betweenness are also the ones who participate the most in Advisory Committee meetings (see Figure 4.2 for a graphical representation). However, the correlation was only statistically significant for betweenness (p=0.000) and not for indegree (p=0.108). Results of Wilcoxon rank-sum test showed the same pattern (Table 4.2). The category that held more centrality was the park employees, but the association was only significant for indegree (p=0.000) and not for betweenness (p=0.672) (Table 4.3). Regarding these results, we can say that central stakeholders of the communication network are represented in participatory bodies. However, the abovementioned lack of trust can have negative effects on the social network. This is why it is interesting to discuss the role these participatory bodies can have in enhancing communication among stakeholders.

Table 4.1 Spearman correlations between "individual centrality" and "assistance to Advisory Committee meetings" (n=198).

	Assistance to Advisory Committee
Indegree	0.115
Betweenness	0.369***

Notes: *Significant at p≤10%, **Significant at p≤5%, ***Significant at p≤1%

Table 4.2 Wilcoxon rank-sum text between "individual centrality" and "assistance to Advisory Committee meetings" (n=198).

	Indegree				**Betweenness**			
	Mean	**SD**	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**
Assistance	2.70	5.03	0	29	279.98***	1139.85	0	8844.74
No assistance	1.55	1.65	0	16	29.53	138.27	0	1008.66

Notes: *Significant at p≤10%, **Significant at p≤5%, ***Significant at p≤1%

Table 4.3 Wilcoxon rank-sum text between "individual centrality" and "working at the natural park" (n=238).

	Indegree				**Betweenness**			
	Mean	**SD**	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**
Park Employees	4.2***	6.10	1	28	227.14	565.84	0	2112.94
No Park Employees	1.60	2.24	0	29	77.22	616.76	0	8844.74

*Significant at p≤10%, **Significant at p≤5%, ***Significant at p≤1%

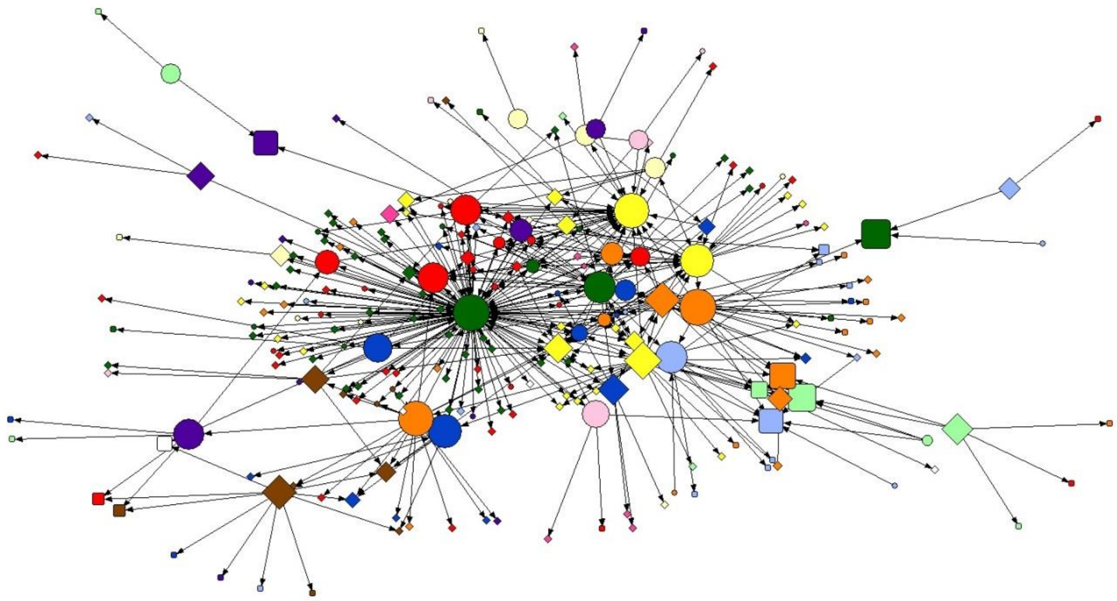

Figure 4.2 Communication network of the natural Park and assistance to the Advisory Committee meetings

Notes: The size of the nodes indicates the betweenness, the shape the assistance to the Advisory Committee meetings (circle for assistance, diamond for not assistance, rounded-square for missing data (i.e. institutions)). In addition, the following colors are used to denote the various categories: Red (Local administration); Green (Park managers); Yellow (Park employees); Blue (Agricultural sector); Orange (Scientific sector); Purple (Civic sector); Pale green (Conservationist sector); Pale blue (Leisure sector); Pale yellow (Environmental education-Tourist sector); Brown (Forestry sector); Pink (Accommodation and restaurants); Pale pink (Other enterprises); White (Others).

At first glance we could say that the Advisory Committee works as a positive feedback loop (Beilin et al., 2013) reinforcing the existence of the social network of communication in the natural park. One of the stakeholders interviewed put it this way *"The advisory committee is a place for exchanging information and knowledge and to meet with each other"*. Beilin et al. (2013) highlight that the attendance to social events underpins community engagement in community based, natural resource management through the strengthening of social networks.

However, if lack of trust and frustration (Hoppe, 2010) are taking place, then a negative feedback loop may be created, which results in some stakeholders dropping out of the network or being placed into the periphery, as a consequence of disempowerment or disengagement with the group's interests (Beilin et al., 2013). One of the farmers interviewed expressed this with the following words: *"The farmers are not any longer going to the meetings as these do not represent our interests and do not work well. We are the land managers, the land custodians and people do not recognize it."* The exclusion of some stakeholders, e.g., agricultural producers, from the Advisory Committee meetings could represent a loss of a central position in the communication network. This might result in losing valuable management land knowledge for the entire network.

Our overall results reveal that the three aspects, i.e. communication, trust and participation, are interrelated. The process of building trust (mistrust) through communication and participation reinforces (weakens) the communication network and effectiveness of participatory bodies. At the same time, participation can reinforce or weaken the communication network and vice versa.

The centrality of park employees in the communication network might be caused by the necessity to maintain internal connectivity centered on certain members in official roles to keep the stakeholders informed about park issues (Beilin et al., 2013). We argue that this form of connectivity could be effective if trust and power work in synergy, obtaining positive effects such as sharing information, providing support, and increasing cooperation (Vollan, 2008; Graham, 2014). However, if legitimate power, i.e., a person's perception that a social agent has a legitimate right to prescribe behavior for him (French and Raven, 1959), acts as substitute of trust, the outcomes can have ramifications throughout social networks with negative consequences for the achievement of collective action (Graham, 2014).

4.3.3 Selection of stakeholders for assessing the adequate governance structure of the natural park

As mentioned above, results from non-participant observation and semi-structured interviews showed a general disappointment with participatory bodies. We used the outputs of social network analysis to select stakeholders because our results also proved that there exists an informal communication network in the natural park and that it is, at least partially, interlinked with formal participatory institutions. Based on measures of indegree and betweenness centrality, network of conflicts between stakeholders, core-periphery and hierarchical clustering analysis, and category of stakeholder, we selected 12 participants for future participatory processes (Table 4.4). These participants represent the various categories of stakeholders identified within the natural park. Prell et al. (2001) reflect on the importance of diversity in considering a wide representation of stakeholders. The person selected from each category was the one who obtained both a higher indegree and betweenness centrality. High indegree centrality guarantees that these actors can motivate the network and rapidly diffuse information through it, and high betweenness centrality assures that actors link disconnected segments of the network because they have a more holistic view of the entire network and have the capacity to mobilize and diffuse information to the larger network (Prell et al., 2008, 2011). Notwithstanding, as the literature notes (Prell, 2011), it is possible that focusing on more central actors does not lead to the best selection of stakeholders. For this reason, we made sure that our selection included people from the core and the periphery of the network and that the four clusters obtained in the hierarchical clustering analysis were represented in our selection. By avoiding homophily, i.e., a situation in which similar actors are attracted to one another and thus choose to interact with each other, we made sure to avoid gathering redundant information (Prell et al., 2010) and to guarantee diversity based on social networks, i.e., diverse positions within a wider network structure (Prell et al., 2011). Because having stakeholders from different backgrounds could lead to conflicts and difficulties in transferring tacit and complex information (Prell et al., 2011), our stakeholder selection also avoided bringing together people with some degree of conflict to achieve a maximally productive assessment of the governance structure of the natural park.

Table 4.4 Stakeholders selected for future participatory processes.

Category	Indegree	Betweenness	Core/Periphery	Cluster	Person
Local administration	7	878.79	Core	4	Mayor of a municipality
Park managers	29	8844.74	Core	1	Park director
Park employees	28	2112.94	Core	4	Park technician
Agricultural sector	4	537.04	Periphery	3	Farmer
Scientific sector	5	158.523	Core	3	Veterinarian of Universitat Autònoma de Barcelona
Civic sector	1	107.500	Periphery	2	Local activist
Conservationist sector	5	0	Periphery	2	Member of a conservationist organization
Leisure sector	4	241.375	Periphery	3	Member of a hiking group
Environmental education-Tourist sector	1	80.37	Periphery	1	Member of a cooperative working on environmental education
Forestry sector	4	392.00	Periphery	2	President of a Forestry Defense Association
Accommodation and restaurants sector	1	114.46	Periphery	1	Manager of a restaurant
Other enterprises	1	114.29	Periphery	3	Environmental restoration manager

4.4 Conclusions

Results from our study suggest that the communication network of Sant Llorenç is fragile because of the few ties between stakeholders, which reflect a lack of trust and little knowledge exchange. Our research also suggests that central stakeholders of the communication network are represented in participatory bodies of the natural park. We argue that social network analysis is an appropriate tool to identify central stakeholders of different categories to support participatory processes. In light of these findings, we consider it important to improve the functioning of participatory bodies and to initiate participatory processes to generate or rebuild trust, share information, provide support, and increase cooperation between stakeholders. Further research could address the question of which network structures are most suitable for local participation in governance (Barabási, 2009; Newman, 2003) to advance the field of social network analysis and natural resource management.

Because effective governance of protected areas has been deemed a priority in developed and developing countries, the implications of our findings are not limited to the natural park of Sant Llorenç del Munt. As Ernstson (2011) noted, we argue that transformative change in natural resource management is critical for the maintenance of social-ecological systems. We suggest that social network analysis, beyond being a supportive tool for stakeholder analysis (Prell et al., 2008), can help to the aforementioned transformation supporting protected areas' policies and governance.

Chapter 5. Unravelling stakeholder participation in protected area governance[12]

5.1 Introduction

Stakeholder participation in biodiversity governance – including civil society organizations, private sector and landowners – is often considered an important factor for promoting both the sustainable management of natural resources and biodiversity conservation (Stoll-Kleemann and O'Riordan, 2002; Dudley, 2008; Borrini-Feyerabend et al., 2013; CBD, 2011). The reason is that stakeholder participation is regarded to increase the social and political acceptance of protected areas potentially contributing to their successful management. In addition, participation has been suggested to contribute to environmental justice[13] by allowing all relevant stakeholders to participate in decision-making processes and thus giving them the right to influence the management of their area (Stoll-Kleemann and O'Riordan, 2002; Brechin et al., 2002; Hajer, 2003; Paloniemi et al., 2015).

In line with this idea, an important motivation for restructuring of biodiversity governance in Europe, especially during the last two decades, has been the argument that effective conservation hinges on consensual decision-making networks involving all relevant stakeholders (Apostolopoulou et al., 2014). This has given rise to various network-based forms of governance and participatory arrangements. However, in practice, at all political levels, from EU to local, there is a considerable gap between the rhetoric of participation in political discourses and actual legal arrangements and policy procedures. Actually, there are rather few cases of genuine participatory processes (Rauschmayer et al., 2009). In fact, these new forms of governance do not always mean enhancing democracy, empowering citizens, and more effective governance (Swyngedouw, 2005). One reason is that rules and norms of participatory processes tend to be established rather top-down on the basis of informal or non-codified principles. Another is that there is often ambiguity on who participates and how. In addition, responsibility and accountability for decisions taken is not always clear. Moreover, as Harvey (2005) argues, this realignment from top-down state governing

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[13] Environmental justice incorporates social justice in environmental issues, which in the context of biodiversity conservation implies three broad principles: (1) the right to participate at all levels of the policymaking process as equal partners, (2) the right to self-representation, and (3) the right to political, economic, and cultural self-determination (sovereignty) (Brechin et al., 2002).

alone to networked governance by a broader configuration of the state-civil society-markets ensemble has been marked under neoliberalism. In the case of environmental governance it has involved a reconfiguration of the institutional arrangements charged with managing nature to favor market-based actors and practices (Bridge and Perreault, 2009). In this shift from ‘top-down’ to ‘bottom-up’ decision-making, ideas such as profitable public–private partnerships and increased business involvement are often promoted. These changes in biodiversity governance are intertwined with what several authors have called “neoliberal biodiversity conservation” (Apostolopoulou et al., 2014; Apostolopoulou and Adams, 2015; Spash, 2015). Examples of neoliberal approaches to conservation in Europe are market-based instruments such as payments for ecosystem services, incentives to attract private sector investment in green infrastructure and biodiversity offsets, and the European Commission’s commitment to develop the Business and Biodiversity Platform (EC, 2011).

A key challenge for biodiversity governance is to support the development of institutional arrangements that can ensure sustainable natural resource management, notably biodiversity conservation, together with promoting environmental justice. In order to advance knowledge about which institutional arrangements can better combine these three aspects, we explore the participatory arrangements established in the natural park of Sant Llorenç del Munt. This park represents a relevant case because the history of its establishment and its current governance dynamics allow us to analyse how participatory arrangements, first introduced in 1986, have transformed power relationships. To achieving our key research aim, we explore three central questions: (1) why was stakeholder participation promoted in the management of the natural park of Sant Llorenç? (2) Which topics are being addressed in the participatory process and in which manner? And (3) who is included in or excluded from the participatory process? We argue that unravelling the political dimension of stakeholder participation in decision-making can contribute to a thorough and critical assessment of participation in the context of an increasing adoption of a neoliberal agenda in biodiversity governance across Europe.

5.2 Theoretical background

We draw on the field of political ecology (PE) of conservation by examining the intersection of socio-economic context, political relations, cultural practices, and ecological processes to highlight how particular environmental governance and management regimes become dominant and how this affects nature-society relations (Bryant, 2001; Walker, 2003; Neumann, 2005). PE further explores how local-scale social dynamics can be understood within broader national, regional and global settings (Robbins, 2002; Walker, 2003). It focuses on how processes of power shape conservation practice and thus its social and ecological outcomes (Roth, 2015). The establishment of participatory arrangements involves sharing of power between the governed and the government, and thus involves various actors and power relations, structural inequalities and different class, ethnic, cultural and gender groups. Participatory arrangements create institutions that empower new actors, while disempowering others (Swyngedouw, 2005). Furthermore, participation as praxis is a terrain of contestation in which relations of power between different actors shape and reshape the boundaries of action (Cornwall, 2008). To understand these processes and dynamics it is necessary to consider the multiple interests and actors within communities, how these actors influence decision-making over time, and the internal and external institutions that shape the decision-making process (Agrawal and Gibson, 1999).

There is a growing literature explicitly linking the emergence of participatory governance forms with neoliberalism (Harvey, 2005) and particularly with a neoliberal shift in conservation (Igoe and Brockington, 2007; Büscher et al., 2012; Apostolopoulou et al., 2014). Büscher et al. (2012) and Igoe et al. (2010) define neoliberal conservation as an amalgamation of ideology and techniques strengthening synergies between growing markets, private investments and effective biodiversity conservation, and the consequent revaluation of nature in economic terms. A key idea is that conservation needs to make 'economic sense', namely by generating economic activity around it.

The role of emerging governance forms, including participatory arrangements, is approached as a political process (Neumann, 2009; Adams and Hutton, 2007; Walker and Hurley, 2004; Apostolopoulou et al. 2014). It means highlighting its political dimension, which is the "space of power, conflict and antagonism within human societies" (Mouffe, 2000). The idea here is that processes of participation take place within existing structures where some groups have more power than others and possess advantages in shaping consensus in favour of hierarchical relationships (Peterson et al., 2005), thus generating dynamics of inclusion and exclusion of stakeholders.

The enhancing of participatory arrangements by a spectrum of organizations (from grass-roots movements to the World Bank) (Swyngedouw, 2005; Cornwall, 2008) raises questions on what really counts as participation. Actually, participation has different meanings as reflected by the three overlapping schools of thought and practice about participation (Pretty, 1995; Bishop and Davis, 2002; Niedzialkowski et al., 2012) resulting in different typologies of stakeholder participation (e.g. Arnstein 1969; Pretty 1995; White 1996; Davidson 1998; Bishop and Davis 2002; Webler and Tuler 2006). Our study draws on this diverse literature.

The first school sees participation as a political process where the main aim is to mobilize collective action, empowerment and institution building. Thus Arnstein (1969) considers that citizen participation is a categorical term for citizen power, meaning that participation without redistribution of power is an empty and frustrating process for the powerless. Arnstein distinguishes between modes of tokenism and modes of citizen power, i.e. citizen participation. The former rely primarily on decision makers informing or seeking public advice on their proposals with no commitment to include their opinions in decision-making, and the later implies giving some level of power to stakeholders.

The second school sees participation as a means to increase social-political acceptance of policies, the central notion being that if people are involved, then they are more likely to agree with and support policies (Bishop and Davis, 2002). Here, public involvement in a decision is not considered as inherently good; rather, its desirability depends on the issue at hand. In some cases extensive involvement is desirable but in others minimal or no involvement is preferable, notably when there is limited social knowledge about the topic. As an example, Bishop and Davis (2002) consider that participation needs to be framed less laden by idealist notions of democracy and more defined by the policy tasks at hand. They propose a set of techniques without relying on a continuum ranging from minimum participation to maximum participation.

Recently, a third school has appeared focused on the deliberative understanding of participation (Renn, 2006). It sees participation as a means to include different knowledges in order to improve the quality of decisions and promote the understanding of such decisions.

These three schools of thought correspond to the three rationales for participation – normative, instrumental and substantive – identified by Fiorino (1990) and Stirling (2006, 2008). Wesselink et al. (2011) added another rationale – legalistic- thus proposing four different rationales for participation in environmental governance: i) instrumental, ii) substantive, iii) normative and iv) legalistic (see Table 5.1). The concept of rationales for participation refers to the systems of justification for participation that determine the characteristics of established participatory arrangements. In our case study, we analyse the political process associated with the participation in the governance of the natural park of Sant Llorenç and its implications. This involves assessing why, what, how and who is involved. In addition, assessing stakeholders' perceptions regarding the drivers of stakeholder involvement by park managers can shed light on the type of participation chosen.

Table 5.1 Rationales for public participation

Rationale	Description of rationale
Instrumental	Effective participation makes decisions more legitimate and improves results. It aims to restore public credibility, diffuse conflicts, justify decisions, and limit future challenges to implementation by "creating ownership". Policy goals are not open for discussion; only the details are. It hereby supports incumbent interests.
Substantive	Non-experts see problems, issues, and solutions that experts miss. It aims to increase the breadth and depth of information and thereby improve the quality of decisions; it ignores power issues (related to problem framing). Policy goals can be changed.
Normative	Democratic ideals call for maximum participation. It aims to counter the power of incumbent interests and allows all who are affected by a decision to have influence.
Legalistic	Participation is only organised to meet formal requirements.

Source: Wesselink et al. (2011).

5.3 Study site and methods

5.3.1 The natural park of Sant Llorenç del Munt

The natural park of Sant Llorenç was initially protected in 1972 by the *Pla Especial d'Ordenació* (Special Land Use Plan) under Franco's dictatorship (1939-1975). As explained in section 1.3, the reason for the initial protection of the area was to legitimize residential areas in the surroundings of the protected area while conserving higher areas (Aguilar, 2012). This threat of quick urban spread in an area with high ecological and social value stimulated the emergence in 1978 of the first environmental movements in Catalonia and the Iberian Peninsula, the "Coordinadora per la Salvaguarda de Sant Llorenç del Munt i la Serra de l'Obac" (hereafter Coordinadora). This movement led by the Coordinadora was mainly structured around hiking groups present in most villages and cities in the surroundings of the protected area and was highly influenced by the political and social context of the democratic transition period after Franco's dictatorship (Aguilar, 2012). This meant that there were demonstrations and collective action in order to ensure the protection of the area. This originated a conflict arising during the implementation of state development and conservation policies as happened elsewhere (see e.g. Apostolopoulou and Pantis, 2010).

The achievements of the Coordinadora in defending the territory were really important to the point that it has been considered a key actor for the protection of the natural park of Sant Llorenç (Aguilar, 2012). The area was finally assigned the official status of a Natural Park by the Catalan government in 1987. Moreover, a participatory body of the park was established in 1986, partly due to the requests of the Coordinadora, in order to ensure the involvement of stakeholders in decision-making of the natural park. After the protection of the area, the Coordinadora continued to be active in defending the conservation of the area, notably after conflicts arose, such as over the construction of an unpaved road through the natural park in 2006, called Matarodona road.

5.3.2 Description of the governance of the natural park

The Natural Park is managed by the Diputació de Barcelona, a regional administration corresponding to the territorial area of the Barcelona Province. The governance of the natural park includes two participatory bodies: the Coordinating Council and the Advisory Committee (hereafter AC).

The Coordinating Council is the formal institution established in 1983 that guaranties the participation and collaboration in park management of the different public administrations with competencies in the area. It meets every six months and it is composed of representatives of the Diputació de Barcelona, representatives from the council of each municipality that has part of its area inside the protected area (Figure 1.1), a representative of the Catalan Government (Generalitat de Catalunya), and a representative of the park's AC.

The AC is an informative meeting of stakeholders and park managers open to the public held every six months. Its main objectives are twofold: 1) Inform stakeholders about the policies and actions implemented or planned to be executed; and 2) collect the comments of stakeholders on the issues presented (even though these comments are not binding). It was established in 1986 in order to "guarantee stakeholder participation, seen as a non-professionalized and unpaid voluntary action, and aimed at facilitating the suitability of decision-making to social demands" (Diputació de Barcelona, 1997). It is composed by representatives of the Diputació de Barcelona, the Coordinating Council, and the various social, economic, scientific, cultural and conservationist organizations with a stake in the management of the natural park.

In 2010, the AC was joined with another participatory body, the Permanent Forum of the European Charter for Sustainable Tourism (ECST) due to a proposal made by park managers and voted and approved in one AC. The ECST is an initiative of the EUROPARC Federation which overall objective is to promote the development of sustainable tourism by establishing a voluntary commitment between the managers of protected areas and stakeholders. This is one of the priorities of the natural park and many resources have been designated to ensure active participation by locals and also to inform and educate local people through workshops and trips. There is a permanently employed staff of the ECST. This participatory body discusses and approves the proposals concerning the ECST and further coordinates actions and initiatives developed by the diverse stakeholders (both individuals and organizations) to joint efforts in their projects.

Our research focuses on analysing the participation in the AC as a formal participatory body, given that this is the body attended by a wide range of stakeholders with a stake in the natural park.

5.3.3 Methods of data collection

Research was conducted in the Natural Park of Sant Llorenç del Munt (Figure 1.1) and its surroundings between January and November 2013 and between April 2014 and January 2015. Our research involved a combination of methods: 1) review of documents relevant to the governance of the natural park of Sant Llorenç and meeting minutes of the AC and the Permanent Forum; 2) non-participant observation; 3) 25 semi-structured interviews with relevant stakeholders of the natural park, 4) a workshop with 11 stakeholders previously selected through a Social Network Analysis, and 5) 11 questionnaires presented to the workshop attendants.

5.3.3.1 Review of documents

In order to assess to what extent participation is part of the natural park's governance, various documents regarding the governance of the protected area were reviewed (e.g. Catalan Law 12/85 on Natural Areas, Decree 328/1992 on Plan for Areas of Natural Interest, Decree 106/87 on Declaration of the Natural Park and the last version of the Special Plan for Protection of the Physical Environment and Landscape of the Natural Park modified in 1997). We further reviewed all the available documents with minutes of the 14 meetings of the AC held from 2008 to 2014. Of these, three pertained to the Permanent Forum, four to the AC and seven to both bodies jointly. We reviewed these documents to assess who participated and with which frequency, and to which category of stakeholders they belong to.

5.3.3.2 Non-participant observation

We used non-participant observation techniques to establish contacts/engage with the community, local culture and local social organization in a non-active way (Bessette, 2004). We undertook several trips from January to June 2013 in order to visit the twelve municipalities that have part of their area within the natural park. This involved informal talks with individuals or groups in order to explore the relationship of each municipality with the natural park. In addition, we stayed two weeks in July and August 2013 in a farmhouse inside the park to obtain first-hand information about the situation of people living inside the protected area and their perceived role of participation in the park governance. We also participated in two meetings of the AC in 2013 and 2015 in order to observe how the participatory body works in practice and how the various stakeholders interact in it.

5.3.3.3 Semi-structured interviews

We conducted 25 interviews with relevant stakeholders concerning participation in the governance of the natural park. We selected stakeholders who had an interest in the park's management because they affect or are affected by decisions regarding it (Reed et al., 2009). Stakeholders were selected on the basis of reputation or recommendation (following a "snowball strategy"). We further verified their relevance through a Social Network Analysis (SNA) to map local exchange network of information between stakeholders conducted in the natural park (see chapter 4). As part of the SNA we performed a *core-periphery analysis* to identify which stakeholders belong to the core and which belong to the periphery of the network. We wanted to have the opinion of actors belonging both to the core and the periphery of the network, so we interviewed 13 core stakeholders and 12 peripheral ones. This involved people from different categories of stakeholders, such as local administrations (mayors and councillors of environment), park managers and employees, representatives of conservationist organizations, and workers in agricultural, scientific, tourism, leisure, education, and forestry sectors.

The interview questions were formulated in a way to obtain clear insights into the respondents' opinions on the participatory body. They were aimed at assessing respondents' views on the usefulness of the participatory body, the reasons the park managers promoted it and respondents' experiences with it. The interviews lasted 1.5 to 2 hours and were all tape-recorded.

5.3.3.4 Participatory workshop

We ran a workshop with 11 previously selected stakeholders to assess participation in the governance of the natural park. The selection of stakeholders was based on the SNA of the communication network of the natural park (see section 4.3.3). SNA has been proven to be an effective and reliable method to select stakeholders for participatory processes (Prell et al., 2011). The stakeholders selected represented the 12 categories of stakeholders present in the natural park. These cover local administrations (mayor and councilors of environment), park managers, park employees, representatives of conservationist, civic and leisure organizations, workers in the agricultural, scientific, tourism, environmental education (its representative could not attend the workshop) and forestry sectors; and other companies related to the natural park. The stakeholders selected are the most central persons in the communication network within their assigned stakeholder category according to the previously done SNA (for detailed information see Calvet-Mir et al., 2015).

We divided the stakeholders into two groups of 5 and 6 participants. Each group participated in deliberative discussions following brief introductions to the research topics. The discussions explored the strengths and weaknesses of the AC considering: i) its effectiveness, ii) its content (the issues addressed in it), and iii) its development (how discussions and decisions are taken, the suitability of frequency and place of meetings).

Selected stakeholders were also asked to provide recommendations for a better functioning of the body. The discussions were facilitated by the first and the second author of the article and two research assistants. Each discussion lasted 30 minutes and all discussions were recorded and reported by taking extensive notes.

5.3.3.5 Questionnaire

We asked the 11 workshop participants to fill in a questionnaire (see Appendix A.5.1) that contained both questions about general participation issues and more specific questions about the AC (e.g. the reasons why they thought the park managers promoted this participatory body).

5.3.4. Methods of data analysis

In order to explore stakeholders perceptions on why stakeholder participation is being promoted by park managers (research question 1), we relied on data from the interviews and the questionnaire and we used the concept of rationales for participation. We selected, separated and sorted out data following Wesselink's (2011) categories of rationales (see Table 5.1). When a respondent presented a combination of rationales these were counted separately, so the total score is higher than the number of interviews/questionnaires. Five people participated both in the interviews and questionnaires, however their opinions have been considered just once (number of respondents = 31). We used descriptive statistics to assess the different types of rationales for participation perceived by stakeholders and we related them to the different categories of stakeholders interviewed.

In order to assess what and how is being addressed in the AC (research question 2), we triangulated (i.e. cross-verified) information from the review of documents, non-participant observation, semi-structured interviews, the workshop and the questionnaires. The information was analysed following the method of qualitative content analysis (Miles and Huberman, 1994). A list of viewpoints was compiled and reorganised by aggregating similar statements regarding the three aspects analysed for the participatory body (content, development and effectiveness) as well as proposals for improving it. The aim of the analysis was not to compare or count the opinions of various participants, but to provide insights for the characterization of the participatory body. To assess the character of the AC, we relied on data from the questionnaire (see Appendix A.5.1). We asked respondents to rank with a scale from zero to five the character of the AC, with zero denoting "it is not" and five "it is a lot". Four characters of the AC were described: 1) informative, meaning that the actions done or planned to be done were explained to participants in a unidirectional way; 2) consultative, meaning that participants' opinions were requested without a joint discussion; 3) interactive, thus engaging in joint discussions among participants leading to new opinions; and 4) decisive, this meaning that the agreements adopted were executed. We calculated the average punctuation of each attribute of the AC, that is, informative, consultative, interactive or decisive. We finally categorized the type of participation promoted in the AC following Pimbert and Pretty's (1997) classification of eight different typologies of participation along an axis of low and high involvement of stakeholders (Table 5.2). We have considered this classification because it is widely used for exploring participation in protected areas.

Table 5.2 A typology of participation.

Type	Components of each type
Manipulative participation	Participation is simply a pretence, with people's representatives on officials boards but who are unelected and have no power
Passive participation	People participate by being told what is going to happen or what has already happened. It is unilateral announcement by an administration or by project management; people's responses are not taken into account. The information being shared belongs only to external professionals.
Participation in information-giving	People participate by answering questions posed by extractive researchers and project managers using questionnaire surveys or similar approaches. People do not have the opportunity to influence proceedings, as the findings of the research or project design are neither shared nor checked for accuracy.
Participation by consultation	People participate by being consulted, and external agents listen to views. These external agents define both problems and solutions, and may modify these in the light of people's responses. Such a consultative process does not concede any share in decision-making and professionals are under no obligation to take on board people's views.
Participation for material incentives	People participate by providing resources (labour, in return for food, cash or other material incentives). Much in-situ research and bioprospecting falls in this category, as rural people provide the resources but are not involved in the experimentation or the process of learning. It is very common to see this called participation, yet people have no stake in prolonging activities when the incentives end.
Functional participation	People participate by forming groups to meet predetermined objectives related to the project, which can involve the development or promotion of externally initiated social organization. Such involvement does not tend to be at early stages of project cycles or planning, but rather after major decisions have been made. These institutions tend to be dependent on external initiators and facilitators, but may become self-dependent.
Interactive participation	People participate in joint analysis, which leads to action plans and the formation of new local groups or the strengthening of existing ones. It tends to involve interdisciplinary methodologies that seek multiple perspectives and make use of systematic and structured learning processes. These groups take control over local decisions, and so people have a stake in maintaining structures or practices.
Self-mobilization	People participate by taking initiatives independent of external institutions to change systems. Such self-initiated mobilization and collective action may or may not challenge existing inequitable distributions of wealth and power.

Source: Pimbert and Pretty (1997).

We used the information from the review of documents, the workshop and the questionnaires to assess the different stakeholders attending the participatory body and their reasons for participating (research question 3). Descriptive statistics were generated to indicate who participated in the body considering stakeholder categories and four different periods of time. This includes attendance to i) the AC between 2008 and 2014; ii) the AC before being joined with the Permanent Forum, iii) the Permanent Forum before being together with the AC; and iv) the last two meetings held in 2013 and 2014. We coded the different arguments stated for participating. To assess the perceived influence and relevance of each category of stakeholders, we relied on data from the questionnaires. We asked respondents to rank with a scale from zero to five, with zero denoting "no influence/relevance" and five "a lot of influence/relevance", the different categories of stakeholders considering their influence in the AC and their relevance in terms of attributed importance by the others in decision-making. We then calculated the average punctuation of the perceived influence and relevance of each category of stakeholders participating in the body.

5.4 Results and discussion

5.4.1 The reasons for promoting participation

Most research participants (51%, 23 opinions over the total of 45) perceived that the prime drivers of stakeholder involvement in the governance of the natural park by park managers were of instrumental nature, namely that participation has been mainly promoted to increase the legitimacy of decisions. As one park employee stated: "*the Advisory Committee is a tool to build complicity and consensus*". The legalistic rationale, namely participation promoted to meet formal requirements, and the substantive rationale, promotion of participation to incorporate different types of knowledge, were also considered as important each making up 20% of the total (9 opinions each one over the total of 45). Thus, for example, one park employee pointed out that participation was enhanced because the Special Plan of the natural park obliged the undertaking of participatory arrangements with stakeholders (legalistic rationale) whereas a mayor of a village argued that "*there are stakeholders that have the solution to the problems faced by the natural park, and the Advisory Committee is a good place to incorporate these solutions to decision-making*" (substantive rationale). Only four opinions referred to the normative rationale, namely the promotion of participation to allow all who are affected by a decision to have influence, corresponding to the 9% of total opinions (Table 5.3). Thus, for example, an employee in the tourism sector argued that "*the Advisory Committee is promoted in order to incorporate people's views in decision-making. People going to the park and people living in it are the ones that best know the challenges of the protected area*".

Table 5.3 Distribution of opinions among rationales for promoting participation and stakeholder categories.

	Rationale			
	Instrumental	**Substantive**	**Normative**	**Legalistic**
Stakeholder categories				
Local administration	4	2		
Park managers	2	1	1	1
Park employees	2	2	1	1
Agricultural sector	4			3
Scientific	2	1		1
Civic			1	
Conservationist	2			2
Leisure	3	1		
Environmental education	2	1		
Forestry	2			
Tourism			1	1
Other companies		1		
Total	23	9	4	9

Notes: Numbers denote how often the respective opinion corresponding to each rationale for participation was expressed by stakeholders. When a respondent presented a combination of rationales these were counted separately, so the number of opinions is higher than the number of respondents. Number of respondents, N=31.

One aspect of the instrumental rationale is to increase the legitimacy of decisions undertaken, thus diffusing conflicts (Table 5.1). In our case study, there has been a reduction of conflicts compared to the conflicts occurring in the 1980s (Aguilar, 2013). However, increasing legitimacy by promoting participation was not the reason for diminishing conflicts. As some interviewees have pointed out the establishment of the AC and its dynamics led to the inclusion in the park governance of stakeholders that were previously active in protecting the area and have been gradually demobilized. They explained that the AC was created upon the request of the Coordinadora and its initial purpose was to counter past management trends that focused on development instead of conservation. The Committee was expected to be composed by environmental organizations, researchers and groups interested in conservation. However, park managers changed its composition by allowing everyone related to the protected area to attend the body. This resulted in an AC composed of people with very diverse interests, such as landowners, hunters and people from tourist sector together with conservationist groups. Interviewees considered that this shift changed the dynamics of the body, making it inoperative because it became too difficult to reach any meaningful consensus. As one of the interviewees from a conservationist organization explained: "*the democratic argument stated by park managers that everyone should participate resulted in diminishing participation because many conservationist* *organizations stopped participating in it*". This shows the limits of consensual approaches that are based on the simplistic argument that social groups with differing interests can find mutually beneficial solutions through participatory processes. In fact, this may also result in processes of self-exclusion of certain stakeholders as happened in our case study. Participation can thus be considered as a governmental tactic of reducing conflicts by including in park governance organizations that have been critical about the management of the protected area.

Moreover, the participatory body in Sant Llorenç has no binding character. As an interviewee from a conservationist organization explained the majority of opinions in the participatory body regarding certain topics were not considered because of the non-binding character of the body, such as not to build the conflicting road of Matarodona in 2006.

The above critiques from the interviewees emphasize the "tokenism character" of certain types of participation: as explained by Arnstein (1969) stakeholders have the perception that decisions have been taken previously and participation is just a tactic to pretend a false involvement of the community in public decision-making and even co-opting the stakeholders that have been critical with previous management decisions. These interviewees further questioned the power of bureaucrats in deciding the composition and development of participatory arrangements which leads to losing government public credibility.

Our case study shows that the establishment of participatory arrangements was mainly a result of the claims made by the Coordinadora and that normative rationales for participation are not considered relevant reasons for promoting participation by park managers. These findings reinforce the point made by Rauschmayer et al. (2009) that when there is a shift towards more participation at the local level, the prime actual driver is in most cases previous conflicts and not normative choices embedded in governance rhetoric. In these situations, two main rationales for participation coexist, the instrumental one focusing on diffusing conflicts among decision-makers and the normative one among stakeholders starring conflicts requesting the chance to influence decision-making. In our case, the rules defining and governing participation are dictated by decision-makers establishing non-binding processes which do not legitimate decisions at the end. Binding spaces of decision-making that would give some level of power to stakeholders need to be established over time in cases where permanent participatory bodies exist. Otherwise, people would inevitably feel frustrated and co-opted by the governance system.

5.4.2 From consultation to claims for more influence

The AC corresponds to participation by consultation following Pimbert and Pretty's (1997) classification. This means that people participate by being consulted about decisions already made and decisions that will be undertaken by park managers and their opinions are not necessarily included in decision-making. With regard to the content of the meetings of the AC, most respondents agreed that the main topics related to the management of the natural park are discussed although not in depth. Most participants believe that discussions should not evolve around the details of already accepted policies but rather around the principles guiding the management of the protected area. They proposed to implement participatory techniques, such as working groups, in order to properly analyse the topics and be able to make proposals. They further considered that the topics of the meetings should be agreed in advance while giving everyone -and not only park managers and employees- the opportunity to propose them and setting the agenda of the meetings.

Regarding the development of the AC, the results of the questionnaire clearly showed the dynamics of the body. Most participants gave the highest score to its informative character (average score of 4.27 on a scale from zero to five) followed by its consultative characteristics (average score of 3.18), i.e. the fact that participants' opinions were requested without any joint discussion. The AC was considered as also having a rather limited interactive aspect (average score of 2.82), i.e. the engagement in joint discussions among participants leading to new opinions was rarely taking place. Finally, the AC was not considered decisive (score of 2.09). The low scores obtained by the interactive and decisive aspects show that discussions between participants to achieve consensus are infrequent and that the agreements adopted are not executed.

It has to be noted that the official documents about the AC define it as a non-binding body, a characteristic of the AC that has been criticized by some stakeholders (8 out of 25) during the interviews. As one interviewee (mayor of a village) pointed out "*people apparently participate because everyone can say what he/she wants, but then nothing is implemented*". However, park managers and most of the employees agreed with the non-binding character of the AC on the grounds that they believed that making decisions is the responsibility of the park managers and not the AC. Making the agreements binding is part of the process of empowering the community but it is rarely achieved because of the resistance of decision-makers to share power (see e.g. Stoll-Kleemann and O'Riordan, 2002; Songorwa, 1999). As discussed in the previous section, the non-binding character of permanent participatory bodies can lead to complaints among stakeholders about their usefulness. Against this backdrop, participation as praxis is not an immutable process but a terrain where the interaction of participants may require reshaping the conditions and type of participation, the "boundaries of action" as Cornwall argues (2008).

Regarding the effectiveness of the AC, all categories of stakeholders considered it effective except the representatives of the agricultural, civic and forestry sector. The reasons were that it gives information regarding the natural park, that stakeholders can make proposals, that it enhances the social network of the protected area because stakeholders share information and worries, and that it legitimates decisions. One of the interviewees (park employee) stated that "*it contributes to better knowing the socioeconomic reality of the natural park*". Moreover, two interviewees (the park employee and one scientist) considered that the AC was representative of the different stakeholder viewpoints regarding park management. Stakeholders who found it ineffective were mainly concerned with its non-binding character that generates frustration among certain stakeholders. In the joint discussion at the end of the workshop, the majority of stakeholders agreed that the informative character of the AC was not enough and that decisions need to be binding. However, this raises some problems, such as: which decisions can be advised by the AC? How does the AC arrive at an advice (e.g. majority voting?).

Our results show that the AC is a structure that is not devoid of power inequalities where park managers are able to control the information by deciding the topics of the meetings, the terms of participation, and whether they will finally implement the agreements.

5.4.3 Who is participating? Unravelling trends towards neoliberal biodiversity governance and conservation

Results from the attendance to the AC (Table 5.4) indicate there has been a significant shift regarding the different categories of stakeholders that are represented in it. Three major trends can be identified in the evolution of the composition of the AC. First of all, representatives of the environmental education and the tourism sectors participate most frequently even though their presence in the AC before joining the Permanent Forum was almost zero. The unification of both bodies in one was a clear tipping point that explains the change in the composition of stakeholders attending the AC. One person from a hiking group pointed out that most people participating in the Permanent Forum have direct personal or economic interests, such as restaurants and companies dealing with tourism. This has changed the dynamics of the AC, which is increasingly co-opted by the dynamics of the Permanent Forum mainly focused on economic interests dealing with tourism. Secondly, the presence of scientific, civic and leisure sectors in the body has diminished over time. Thirdly, the agricultural, the conservationist and the forestry sectors have completely stopped attending the participatory body.

During the workshop, the representatives of the agricultural and conservationist sectors attributed their decision to stop participating to the fact that the body does not represent their interests and that it has become ineffective. In the case of the agricultural sector, the representative explained that park managers do not support their proposals, like the agreement to set up a farmers negotiating table to enhance organic production and coordination among them. Meanwhile, the conservationist sector explained that their trust in the AC diminished since the conflict arose about the construction of the road of Matarodona in 2006.

If participatory bodies in protected areas are not based on serious efforts by park managers to understand local motivations, governance processes may give the superficial appearance of engagement and legitimacy whilst minimise the potential for those with conflicting views to be given a meaningful hearing (Allmendinger and

Haughton, 2012). As Allmendinger and Haughton (2013, p. 7) explain, the flip side of allowing communities to play a greater role in identifying and addressing local needs is a significantly reduced role for the state 'in favour of a plurality of localist interventions which, despite the rhetoric, is tightly circumscribed'.

Table 5.4 Attendance to the AC and Permanent Forum through time by different categories of stakeholders.

Different periods	**Local administration**	**Park managers**	**Park employees**	**Agricultural**	**Scientific**	**Civic**	**Conservationist**	**Leisure**	**Environmental education**	**Forestry**	**Tourism**	**Other companies**
AC_0	21	9	4.5	3	10.6	16.7	7.6	19.7	1.5	6.1	0	0
Permanent $Forum_0$	22.7	4.5	6	4.5	7.6	12.1	3	6	15.1	3	15.1	0
AC-Permanent $Forum_f$	22.9	10.4	4.2	0	2.1	6.2	0	10.4	14.6	0	29.2	0
Average AC-Permanent Forum	21	8.4	3	3	5.3	13	4.6	13	7.6	6.1	13.7	0.8
Tendency		+	-	--	--	--	--	--	++	--	++	+

Notes:

Results are expressed as percentages relative to the total number of different persons attending the meetings. When a person attended to more than one meeting of the same period of time, it was counted just once. We considered four different periods of time:

AC_0: attendance to the AC before being together with the Permanent Forum [66 different persons in 4 meetings].

Permanent $Forum_0$: attendance to the Permanent Forum before being together with the AC [66 different persons in 3 meetings].

AC-Permanent $Forum_f$: last meetings of the AC and Permanent Forum together [48 different persons in 2 meetings].

Average AC-Permanent Forum: average attendance to the AC from 2008 to 2014, since 2010 together with the Permanent Fprum [131 different persons in 11 meetings].

The last row shows the tendency of changes in attendance by the different categories of stakeholders to the participatory body. ++ means considerable increase of the presence of the category in the body; + means increase; - means reduction; -- means considerable reduction.

The respondents to the questionnaire were asked to express which categories they considered to have more influence and more relevance in the AC. The results (Table 5.5) showed that the categories of stakeholders considered to have high influence are park managers and local administrations. The ones considered to have little influence are the agricultural sector, other companies and the forestry sector.

In the case of the perceived relevance, the categories considered to have high relevance in attending the AC are the scientific sector, park managers, forestry, agricultural and conservationist sectors. The categories having little relevance are other companies, tourism sector and civic sector. Therefore, the stakeholders who are considered more relevant are not actually the ones who more frequently attend the meetings of the AC. On the contrary, the tourism sector despite being considered of little relevance it is one of the most represented categories.

Table 5.5 Influence and relevance of each category of stakeholders in the AC

	Local administration	Park managers	Park employees	Agricultural	Scientific	Civic	Conservationist	Leisure	Environmental Education	Forestry	Tourism	Other companies
Influence	4.2	4.9	3	2.1	3.5	2.9	3.6	3.1	2.8	2.5	2.6	2.25
Relevance	3.64	3.82	3.36	3.73	4.36	3.27	3.73	3.36	3.55	3.82	3.1	3

Notes: Results are the average punctuation (on a scale from zero to five) of the perceived influence and relevance of each category of stakeholders participating in the AC. Number of respondents, N=11.

The fact that the most relevant stakeholders have stopped attending the AC is indicative of the weaknesses of this heterogeneous participatory body. This process of self-exclusion is leading to a progressive hegemony of the private sector represented by small-scale companies related with tourism and environmental education. It reveals that the functioning of this participatory body has resulted in an increase presence of market-based actors rather than being a tool that empowers the community through its active participation in the park management. Some stakeholders pointed out that they feel uncomfortable with this emerging hegemony. For instance, the conservationist sector, even though stopped attending the AC, considered that their presence was needed again in order to counteract the economic interests of companies. As pointed out by Lane (2003), uncritical engagement of actors in decision-making can lead to the development of privatized, corporatist agreements that fail to reflect diverse values and interests. Corporatist agreements usually reflect the interests of one or two non-state actors, rather than the full array of them. This shows the contradictions often present in the discourse of decision-making based on consensus among stakeholders which is that along with the wider inclusion of business sector, other actors (in most cases local community groups) are being increasingly excluded from decision-making. In thinking about who should be in a participatory body, a distinction can be made between actors representing self-interests and those representing normative claims to justice or democracy (Young, 1990) which in the case of protected areas can be claims in favour of conservation and collective interests. In fact, a person from a hiking group reflected on this idea suggesting that not all categories of stakeholders should attend the body. He pointed out that "*if we are talking about a natural park, what does a motorcycle organization have to say about it? The participatory body should be focused on conservation*".

The increasing favouring of private companies in conservation is further happening in the Barcelona regional network of protected areas – which Sant Llorenç is part of. Some stakeholders explained that the authority in charge of the regional network wants to change the governance model from a government-based one to a consortium of municipalities including the outsourcing of several services. Actually, there are two models of governance of protected areas in the Barcelona regional network: 1) the ones that are directly managed by Diputació de Barcelona, and 2) partnerships composed by the Diputació de Barcelona and municipality councils together with other public entities like “Àrea Metropolitana de Barcelona” and private entities like Abertis Foundation[14]. These new forms of governance where the role of non-state actors is expanded through various public-private partnerships and the active participation of public, private, and civil actors clearly reflect a neoliberal type of biodiversity governance (McCarthy, 2006; Corson, 2010; Igoe et al., 2010; Apostolopoulou et al., 2014).

These changes in governance are not coincidental. The adoption of the ECST shows that often the goal of ‘participatory’ or ‘collaborative’ arrangements is to support and promote ‘innovation’ and entrepreneurship (see Harvey, 1989) of small companies within or near protected areas. The enhancement of entrepreneurship in rural spaces is increasingly been framed as the most viable solution for conservation to succeed in an era of austerity by generating economic activity around protected areas and a way to attract donations and capital investments (Harvey, 2005; Apostolopoulou et al., 2014). It further links with the increasing discourse that conservation, to be useful, needs to make economic sense by being places where economic activities are generated. As illustrated by McAfee (1999), Duffy and Moore (2010) and Rytteri and Puhakka (2009), proponents of a more intensive utilization of natural resources usually claim that the legitimacy of nature conservation is essentially based on economic opportunities created for the tourism industry. Ecotourism has been conceptualized as a key example of neoliberal conservation due to its emphasis as a means of achieving economic growth, community prosperity and biodiversity conservation (Igoe and Brockington, 2007). In this sense, it shows how in order for natures to be ‘saved’, conservation is brought to the markets and private investment (Büscher et al., 2012). Market inclusion as an equal partner in conservation is not only related to the creation of new ecological commodities, like marketized biodiversity offsets and conservation credits as is happening in UK (Apostolopoulou and Adams, 2014), but also the further marketization of protected areas through tourism (Apostolopoulou et al., 2014).

These processes are also occurring in an upward scale such as in the implementation of the EU Natura 2000 network. Ferranti et al. (2014) reported a progressive shift regarding the socio-economic stakeholders involved in Natura 2000 network – decrease of attendance of farmers and foresters to an increase of industry and business representatives. One of the reasons explaining this shift is the capacity of the latter to be potential financers of the management of the network. This reveals the increasing neoliberal approach to nature conservation focused on financial issues

[14] Abertis Foundation is member of the partnership managing a protected area called “Parc del Foix” since 2010, together with the Diputació de Barcelona and two municipality councils. Inside the park there is the Castle of Castellet, headquarters of the Foundation, which has been designated as the centre for the Mediterranean Biosphere Reserves of UNESCO. Abertis is the leading International group in the management of motorways.

enhancing the potential economic benefits generated by the network, the economic opportunities of financing it and the use of concepts such as partnership and investment (Ferranti et al., 2014).

In an era of austerity and public sector retrenchment, the proliferation of market-based actors in biodiversity governance goes hand in hand with a reduction in public budget. Actually, some stakeholders highlighted that the authority in charge of the Barcelona regional network of protected areas progressively bets for conceptualising natural parks as economic self-sufficient entities which should have very limited support from public budget and resources for their maintenance. One park employee argued that "*this might imply making visitors to pay for some services or enhancing activities that generate income*". More severe processes are occurring in protected areas managed by La Generalitat, the regional government of Catalonia[15]. From the period comprising 2011-2014, there was a reduction of the public budget for natural parks of 49 to more than 60% depending on the protected area (Gepec, 2015). These cuts have reduced or even eliminated actions to conserve biodiversity, the structure of park employees, forest fire prevention efforts, among others[16]. Conservationist organizations pointed out this shift in the management of Catalan protected areas characterized by an emphasis on the 'economic profitability' of them, the creation of innovative financing mechanisms and the private sector involvement in them (EA, 2012; Gepec, 2015). The approved Strategic Plan for Catalan protected areas 2015-2020 (DAAM, 2015) considers that due to the current situation of austerity, protected areas need to be understood as opportunities and not limitations for the territory. It conceptualizes protected areas as places with the aim of attracting both tourism and investments to create socioeconomic activities and promote growth to revitalize these areas. In this sense, conservation should not be a limitation for exploiting natural resources in them. This situation reinforces the establishment of public-private partnerships in which state and business interests collaborate closely to promote activities aiming to enhance economic resources in protected areas (Apostolopoulou et al., 2014). The above developments reflect the increasingly dominant neoliberal conservation discourse and practice (Igoe and Brockington, 2007; Büscher et al., 2012; Apostolopoulou et al., 2014).

[15] In Catalonia, protected areas located in the Barcelona province are managed by the Diputació de Barcelona (a regional administration) and the rest of them by La Generalitat, which is the Catalan government.

[16] Due to the severe reduction of personnel and budget in protected areas managed by the Generalitat of Catalonia, several directors of the natural parks sent a letter to the Catalan president denouncing it. Relevant scientists, universities, entities and public personalities of Catalonia further signed a declaration denouncing the dismantling of conservation policies in Catalonia (available in: https://defensapatrimoninatural.wordpress.com/declaracio/).

5.5 Conclusions

In this paper, we have explored the political implications of participatory arrangements in the governance of the natural park of Sant Llorenç del Munt i l'Obac, in Catalonia (Spain). Among others, we have assessed the reasons for stakeholder participation, what issues are addressed in the participatory body, and the specific procedures and inclusiveness characterizing the body. The history of Sant Llorenç's establishment and its current governance dynamics have shown how the establishment of participatory arrangements instead of manifesting a shift towards more 'democracy' has actually led to the exclusion of social actors with key roles in the management of the protected area. It has further facilitated neoliberal biodiversity governance by favouring the inclusion of market-based actors.

In particular, regarding the reasons for promoting stakeholder participation, we have demonstrated that participation is mainly enhanced by park managers in order to legitimate their decisions whereas promoting participation to allow affected stakeholders to influence decisions it is almost not considered as a reason for promoting participatory arrangements. Our results show that very often the prime driver for establishing participatory arrangements at the local level is the willingness to superficially cover or depoliticise previous conflicts by promoting the inclusion of organizations that have been critical about its management in governance and not normative choices embedded in governance rhetoric. This happened in the 1978 conflicts in the natural park of Sant Llorenç over the protection of the area and the urban development within and around its borders. Interestingly, there were the stakeholders involved in those conflicts – mainly conservationist and hiking groups from villages and cities surrounding the area – who initially demanded the establishment of participatory arrangements in the park. Two main motivations for participation might have driven this development, namely those of decision-makers to focus on minimizing conflicts, and those of stakeholders to influence public decision-making. The current performance of the AC reveals that park managers are the ones who control its functioning, namely by deciding the topics of the meetings, the terms of participation and the implementation of decisions. Moreover, decisions are not binding and discussions mainly address the technicalities of already accepted policies.

Our research reveals that participatory arrangements can actually lead to the exclusion of social actors with a key role in the management of protected areas instead of contributing to more 'democracy'. In particular, the difficulty of reaching consensus among stakeholders attending the participatory body and the non-binding character of agreements resulted in withdrawal of agricultural, conservationist and forestry sectors from the participatory process. These findings confirm that participation is a political process, and suggest the need for research evaluating, and if necessary reshaping, the conditions of participation. The fusion of the AC with the Permanent Forum of the ECST – a policy instrument mainly focused on tourism – further facilitated a neoliberal approach to biodiversity governance by favouring the inclusion of actors with direct personal or economic interests, such as restaurants and companies dealing with tourism. The dominance of these stakeholders in decision-making might reinforce the framing of protected areas in terms of economic profitability. This goes together with ideas aiming for natural parks to become rather self-sufficient entities, able to generate their own economic resources and depend little on general public budgets. The latter is characteristic of a neoliberal approach to biodiversity conservation which has been suggested to take place in other EU countries, namely in some cases in the UK and Greece (Apostolopoulou and Adams, 2015). In these cases, this approach has been related with a wave of deregulation, decentralization, privatisation and marketization of protected areas and natural resources after the 2008 financial crash. Further research is needed to evaluate the consequences of this new approach in terms of sustainable natural resource management, notably biodiversity conservation.

Appendix A.5.1 Structure of the questionnaire

The questions addressed to the research participants covered the following seven topics:

1. Whether they agreed (or not) with the category of stakeholder to which we have assigned them.
2. The reasons why they participated in the AC of the natural park, both their individual reasons and the usefulness to participate regarding their category. In case they were not participating, they were asked whether they thought it could be useful to participate.
3. The reasons why they thought the park managers promoted the establishment of the participatory body.
4. Whether they thought the participatory body influenced positively the management of the protected area.
5. Whether they thought the body was effective, the reasons of its effectiveness and the issues that needed to be addressed to increase its effectiveness.
6. To rank with a scale from zero to five the character of the AC, with zero denoting “it is not” and five “it is a lot”:
   - Whether the body was informative, meaning that the actions done or planned to be done were explained to participants in a unidirectional way.
   - Whether it was consultative, meaning that participants’ opinions were requested without a joint discussion.
   - Whether it was interactive, meaning engaging participants in joint discussions leading to new opinions.
   - Whether it was decisive, meaning that agreed recommendations were implemented.
7. To rank with a scale from zero to five, with zero denoting “no influence/relevance” and five “a lot of influence/relevance” the different categories of stakeholders considering:
   - Their influence in the AC.
   - Their relevance in terms of attributed importance by others participants in decision-making.

We have to point out that the four first questions of the above list have been presented to the research participants prior to the fieldwork through email. This was done to ensure that their answers would not be affected by the presence of other stakeholders.

Chapter 6. Analysis of policy instruments for biodiversity conservation: the case of the European Charter for Sustainable Tourism[17]

6.1 Introduction

Different types of policy instruments can be designed to conserve biodiversity. Policies may aim to provide prohibitions, barriers, standards (e.g., land tenure and use rights), or incentives like prices (subsidies, land or product taxes, access fees) to alter behaviour and projects that negatively affects biodiversity (TEEB, 2010a). Three major categories of policy instruments have been widely used in the literature (e.g., Michaelis, 1996; Gunningham and Young, 1997; Sterner, 2003), named direct regulation (or command-and-control), economic or price-based instruments, and information provision (also known as moral suasion). Direct regulation includes such instruments as permits, standard-setting and zoning. Economic instruments cover cap-and-trade systems, environmental taxes, charges and fees that put a price on environmentally damaging behaviour, or payments for ecosystem services and ecological fiscal transfers that reward conservation enhancing behaviour. Finally, information and motivational instruments aim to improve the information bases upon which people act, or even shift individual or community preferences towards more conservation. The most important biodiversity policy instruments are summarized in Table 2.2 that also describes the incentive related to the relevant instruments as well as the type of behaviour incentivized. These instruments deal with particular causes of biodiversity loss, such as hunting, habitat destruction (e.g., deforestation, fragmentation), water use (e.g. causing desiccation), and environmental pollution (with climate change as a special and very important case).

The combination of policy instruments, which has evolved to influence the quantity and quality of biodiversity conservation and ecosystem service provision in public and private sectors, is known as a policy mix (Ring and Schröter-Schlaack, 2011).

[17] This chapter is part of a deliverable of the European project CONNECT “Linking biodiversity conservation and ecosystem services”: Maestre-Andrés S., Crouzat E., Koetse M.J., Nordén A., Völker M. 2016. A synthesis report of case studies results on the analysis of policy instruments and governance structures for biodiversity conservation.

Therefore, several instruments from the categories mentioned above can often be found in combination. Some instruments may have been introduced on purpose to enhance the outcome of another instrument. For example, informational instruments are often introduced to provide the knowledge necessary to enhance the outcome of regulatory or economic instruments to relevant stakeholders. In other cases, economic instruments are introduced to reduce the costs of regulation, such as financial support programmes related to protected areas. However, certain instruments can jeopardise the objectives or effectiveness of other instruments because of unwanted interactions, notably antagonism (Ring and Schröter-Schlaack, 2011). In relation to biodiversity and ecosystem services, where multiple problems and objectives are present, Gunningham and Young (1997:286) suggest that "the number of instruments must be sufficient to accommodate each level of biodiversity and the web of institutions acting to conserve it". Each threat to biodiversity and each objective would require at least one instrument. This is in line with the well-known Tinbergen rule for policy instruments, which says that the number of policy instruments must equal the number of policy targets (Tinbergen, 1952). It serves as a guideline for defining an effective policy mix. To be effective, the type of instrument or combination of instruments being applied needs to fit to the institutional context of application. For example, one cannot introduce market-based instruments in a local economy that consists mainly of barter trade or reciprocal relationships.

For addressing effectiveness of biodiversity policy well, the analysis of such policy needs to consider unintended, unwanted and avoidable indirect effects. We think this is an important aspect that is missing in most writings on biodiversity policy. In order to deal with this, section 2.5 proposes the term rebound to make reference to these indirect effects and identify five categories of it, biodiversity (two types), ecological, environmental and service rebound. These terms reflect that certain strategies aiming at conserving specific biodiversity have unintended effects which partly undo the direct conservation benefits, causing them to be less effective than is possible. These rebounds can be reduced by appropriate design of policies and strategies. It is evident that effective biodiversity policy requires the minimization of these rebound effects.

The aim of this chapter is to assess the European Charter for Sustainable Tourism (ECST), a policy instrument implemented in the natural park of Sant Llorenç del Munt. The ECST is a voluntary instrument aiming to promote sustainable tourism by establishing a voluntary commitment between the managers of protected areas and relevant stakeholders. The analysis of the policy instrument is focused on the performance of several criteria, such as effectiveness, efficiency, equity and legitimacy. This chapter further compares the performance of the ECST with different policy instruments of different case studies located in four other European countries in order to develop policy recommendations. It further aims at providing arguments for broadening the analysis of biodiversity policy design by considering various types of rebound effects and illustrates examples of such rebounds from the different case studies analysed. Ultimately, this may give rise to distinct and new views on effective policy options and instruments.

6.2 Framework for analysing instrument mixes for biodiversity policies

Building on a review of policy mix frameworks, Ring et al. (2011) suggested a three-step approach for assessing instruments in policy mixes for biodiversity and ecosystem governance:

Step 1. Identifying challenges and context
When it comes to analysing policy mixes, the focus is not on maximising the effectiveness or efficiency of individual policy measures but on the complementarity of the instruments involved, their interplay and the ability of the policy mix to address all drivers of the underlying problem. The appropriate mix of instruments and actors will hence depend upon the nature of the environmental problem, the target groups and wider contextual factors (see Gunningham and Grabosky, 1998). Against this background, the first step of the proposed approach consists in gaining a thorough understanding of the policy object, i.e. biodiversity conservation and ecosystem services management.

Step 2. Functional role evaluation: Identifying gaps and choosing instruments for analysis
There are three main factors that influence the composition of the mix and that define the functional role of different instruments within the policy mix, namely the performance (and composition) of the existing policy (mix), the context-specific strengths and weaknesses of the individual instruments and lastly the interaction of the instruments within the policy mix.

Step 3. Policy evaluation and design
The characteristics of and challenges associated with biodiversity conservation and ecosystem service management will in many instances require the simultaneous use of multiple instruments. And whenever more than one instrument is implemented, the interaction of instruments is of fundamental importance for overall performance of the policy mix. Against this background, the overall aim of instrument evaluation and design is shifted towards the interaction of instruments in a mix and to what extent instruments are complementary or can create synergetic effects.

To develop policy recommendations we refer to the traditional evaluation criteria while moving beyond the core criteria of effectiveness and efficiency in economic analyses, and extended above into a list of 9 assessment criteria (see Table 6.1). All of these aspects are highly context-specific and so are the methods from various scientific disciplines needed to derive some concrete recommendations.

Table 6.1 List of assessment criteria

Criterion	Definition	Question explored
Effectiveness	Realization of the direct aim of the policy instrument which is not necessarily biodiversity protection; and the contribution to biodiversity protection (environmental effectiveness).	Is the direct aim achieved? Does the instrument have positive effects on biodiversity protection?
Efficiency	Highest net welfare gain, or lowest net financial cost achieved by the instrument	Is the instrument cost-effective?
Equity	We considered three types of equity based on Calvet-Mir et al. (2015): a) equity in terms of access, i.e. local people's ability to participate in the ECST; b) equity in terms of decision making, i.e. participants' perceived fairness in ECST decision-making procedures; c) equity in terms of outcome, i.e. focused on the impact and distribution of ECST outcomes.	Who is participating in ECST? Are the ECST decision-making procedures fair? Is there a fair distribution of the outcomes of ECST among different stakeholders?
Legitimacy	Stakeholder conformity to the process of implementation and to its substance-content	Does the instrument appear legitimate to most stakeholders, regarding both its process of implementation and its content?
Complementarity of policies on one or multiple criteria	Mutual reinforcement of various policies on one or multiple criteria, according to different perspectives: space, time, sectors, public target, and sequencing	Is the instrument complementary to other instruments in the policy mix? Does this combination facilitate the achievement of their objectives?
Overlap and/or conflict between policies	Redundancy causing either a dilution of the effects of one instrument by another (negative overlap) or enhancing mutual effects (positive redundancy) Conflicts between the objectives of different instruments	Does the instrument overlap with other policies in a policy mix? Is it beneficial or harmful to the overall effects? Does the instrument conflict with others?
Monitoring and control	Process implemented to ensure that the instrument is applied or that its objective is achieved.	Is there a monitoring and control mechanism? Is it cost-effective?
Creation of incentives	Motivation for agents to alter their behaviour, e.g. coercion, payment, contract or avoiding a fine/tax.	What causes stakeholders to change their behaviour?
Consistency with institutional-cultural context	Good articulation of the specific institutional and cultural context; related to political and administrative feasibility of practical implementation	Does the instrument seem adapted to its cultural and institutional context?

A typology of rebound of biodiversity policies

This section aims to draw attention to a kind of government or policy failure, namely rebound effects of biodiversity policies that reduce their effectiveness. Such rebound effects are unintended, avoidable and usually unobserved indirect consequences of a policy. A good understanding of such rebound will allow for a better design of effective policies. The types of policies summarized in Table 2.2 are likely to score differently in terms of rebound and thus effectiveness, depending on the context and application. A biodiversity policy can be considered to be effective if it will produce the conservation benefits as desired ex ante (Doremus, 2003). One might also define effectiveness in a more abstract way as attaining the highest marginal environmental benefit associated with a given instrument (Ring and Schröter-Schlaack, 2011). Biodiversity policies can have a number of unintended, unwanted and avoidable rebound effects. If they would not be avoidable it would not make sense to bother about them, although one could then perhaps think of compensation measures.

We propose the five types of rebound, namely Biodiversity rebound I (spatial spillover), Biodiversity rebound II (incongruence between protection of different types of biodiversity), Ecological rebound (changes in biodiversity may lead to various responses in ecosystem functioning), Service rebound (trade-off between biodiversity and ecosystem services), and Environmental rebound (biodiversity policy can generate a negative impact on certain environmental indicators). For definition and clarification of these rebounds see section 2.5.

6.3 Study site and methods

6.3.1 The European Charter for Sustainable Tourism implemented in the natural park of Sant Llorenç

The ECST in Protected Natural Areas is an initiative of the EUROPARC Federation, which seeks to implement the concept of sustainable development in some of Europe's most treasured places. The overall objective of ECST is to promote the development of sustainable tourism by establishing a voluntary commitment between the managers of protected areas and relevant stakeholders. In order to ensure sustainable tourism development, the protected area organises public consultation meetings, and sets up a permanent forum or equivalent arrangement between all those directly concerned: the protected area authority, local municipalities, conservation and community organisations and representatives of the tourism industry. The protected area thus facilitates co-operation and the sharing of responsibilities in order to improve the effectiveness of its mission to protect the environment (Europarc Federation, 2007). The ECST covers three stages:

i) To assess the requirements of the area for sustainable tourism and elaborate and implement a medium-term strategy (5 years) for its development; this is foreseen to take the form of an Action Plan in partnership with local tourism representatives, other business sectors, local people and authorities.

ii) To enable tourism businesses working together with the protected-area authority to receive individual recognition as Charter signatories. This entails a diagnosis by the respective business of all its activity, i.e. the gap between what is offered and what visitors expect, and measures to enhance the local heritage.

iii) To involve tour operators who incorporate the principles of sustainable development into their tourism product, and who organize tours to protected areas. They have to analyse the compatibility of their products with the objectives of the area and make sure that the visitor flow they attract does not lead to a destruction of the heritage resources.

The natural park of Sant Llorenç has already implemented stages i and ii. Regarding stage i, an Action Plan was designed as composed of eight different strategies dealing with several ecosystem services and biodiversity conservation.

The governance system of the ECST is composed of two participatory bodies, namely the Working Group and the Permanent Forum (PF). The Working Group is an executive body responsible for establishing the strategy and concrete actions regarding sustainable tourism. It is formed by the park managers and representatives of stakeholders, and on average attended by no more than fourteen people. The PF is a participatory body that discuss and approves the proposals of the Working Group and further coordinates actions and initiatives developed by the diverse stakeholders (both individuals and organizations) to join efforts in particular projects. It consists of representatives of the Diputació de Barcelona (the regional administration that manage the protected area), representatives from the council of each municipality that has part of its area inside the protected area, and the various social, economic, scientific, cultural and conservationist organizations that desire to be engaged in the Charter.

6.3.2 Methods of data collection

Research was conducted in the natural park of Sant Llorenç and its surroundings between January and November 2013 and between December 2014 and March 2015. It involved a combination of qualitative and quantitative methods: 1) review of relevant documents of the ECST and meeting minutes of the Working Group and PF; 2) non-participant observation; 3) 25 semi-structured interviews with relevant stakeholders of the natural park, 4) a workshop with 11 stakeholders previously selected through a Social Network Analysis, and 5) a questionnaire conducted among workshop attendants.

6.3.2.1 Review of documents

In order to analyse the ECST, various documents regarding it were reviewed, such as documents of the Europarc Federation dealing with the Charter, documents of the ECST of Sant Llorenç and its implementation, minutes of the 15 meetings of the Working

Group and the 10 meetings of the PF held from 2010 until 2014. We reviewed these documents to assess the topics of the meetings, who participated, with which frequency and to which category of stakeholders they belong to.

6.3.2.2 Non-participant observation

We participated in two meetings of the PF in 2013 and 2015 in order to observe the dynamics of the participatory body including the topics addressed in it and how the different stakeholders interact in it.

6.3.2.3 Semi-structured interviews

We conducted 25 interviews with relevant stakeholders that were selected following the method described in section 5.3.3.3. The interview questions were formulated broadly in order to assess respondents' views on the main policies implemented in the natural park and their usefulness. The interviews lasted 1.5 to 2 hours and were all tape-recorded.

6.3.2.4 Participatory workshop

We undertook a workshop with 11 stakeholders to assess the ECST. Stakeholders were selected in the same way as described in section 5.3.3.4. The selected stakeholders represented the 12 categories of stakeholders present in the natural park, which are described in the section of chapter 5 mentioned above. We divided the stakeholders into two groups, each including 5 and 6 participants. Each group participated in two deliberative discussions, following brief introductions to the topics. The first discussion explored the strengths and weaknesses of the ECST and the second one the effectiveness, development and content of the PF. The discussions were facilitated by the first author and three other researchers pertaining to the same research institute of the first author. Each discussion lasted 30 minutes and all discussions were recorded and reported by taking extensive notes.

6.3.2.5 Questionnaire

We asked the 11 workshop participants to fill in a questionnaire (see Appendix A.6.1) that contained both general questions about the ECST and more specific questions about the PF.

6.3.3 Methods of data analysis

In order to analyse the effectiveness of the ECST, we relied on data from the review of documents, semi-structured interviews, the workshop and the questionnaires. The information was analysed following the method of qualitative content analysis (Miles and Huberman, 1994). A list of statements was compiled considering the effectiveness of the overall ECST and the implementation of the selected strategies of the Action Plan. In order to explore the equity of access of the ECST, we used information obtained with the review of minutes of the meetings of the Working Group and the PF to assess the different stakeholders attending the participatory bodies. Descriptive statistics were generated to indicate who participated in the body considering stakeholder categories and two periods of time: i) attendance to the first three meetings of the PF, and ii) attendance to the last two meetings of the PF. In order to address equity in the process of decision-making, we assessed the perceived influence and relevance of each category of stakeholders relying on data from the questionnaires and non-participant observation. We asked respondents to rank with a scale from zero to five – with zero denoting "no influence/relevance" and five "a lot of influence/relevance" – the different categories of stakeholders considering their influence in the PF and their relevance in terms of attributed importance by the others in decision-making. We then calculated the average punctuation of the perceived influence and relevance of each category of stakeholders participating in the body. To assess equity in outcome, we relied on information collected through semi-structured interviews and the workshop, which was analysed following the method of qualitative content analysis. In order to analyse the legitimacy of the ECST, we used data from the workshop and the questionnaires. As part of the legitimacy, we assessed stakeholders' perceptions about the dynamics of the PF relying on data from the questionnaires. We asked respondents to rank with a scale from zero to five the character of the PF, with zero denoting "it is not" and five "it is a lot". Four characters of the PF were described: 1) informative, meaning that the actions done or planned to be done were explained to participants in a unidirectional way; 2) consultative, meaning that participants' opinions were requested without a joint discussion; 3) interactive, meaning joint discussions among participants leading to new opinions; and 4) decisive, meaning that any agreements adopted were executed. We calculated the average punctuation of each attribute of the PF, that is, informative, consultative, interactive or decisive. The rest of criteria were analysed relying on data collected through the review of documents.

6.4 Results and discussion

6.4.1 Identifying challenges and choosing instruments for analysis

Mediterranean protected landscapes are under considerable pressure, especially when close to urbanized areas. In our case study, the natural park is located in the Barcelona Metropolitan Region, which implies an intense land-use and land degradation in surrounding areas and high frequency of visitors coming to the protected area. Moreover, traditional activities have nearly disappeared (e.g., exploitation of oak to produce charcoal) or have greatly diminished (e.g., agriculture, logging and cattle grazing) (Aguilar 2012). Currently the service sector is the largest activity in the park, mainly because of its close relation with tourism. Moreover, there are new land uses appearing, notably associated with particular leisure activities such as hiking and climbing. The increased appreciation of landscapes as leisure spaces might turn them into merely a décor for spending free time, (Buijs et al., 2006), reducing the multifunctionality of such landscapes.

Considering the above mentioned challenges, we analysed the ECST focusing on the strategies of the Action Plan dealing with biodiversity conservation, tourism and agriculture, which are the five strategies listed below. The other three strategies (3, 4 and 8) were not analysed because they were not dealing with these topics.

Strategy 1. Cooperation and participation of different stakeholders of the natural park.
Strategy 2. Monitoring and evaluating the natural and cultural heritage.
Strategy 5. Valorisation of natural and cultural heritage through creating tourism products.
Strategy 6. Promotion of education and knowledge about the natural park.
Strategy 7. Promotion and development of the local economy.

6.4.2 Evaluation of the ECST regarding the selected criteria

6.4.2.1 Effectiveness

The first stage of diagnosing the problems and challenges of the natural park was done in 2010. The Action Plan was elaborated considering this diagnosis and has been implemented from 2010 until 2015. Overall, we find that the ECST has been effective and its implemented actions are focused on improving certain issues related with many ecosystem services, such as the promotion and valorisation of cultural heritage, enhancement of environmental education, and promotion of sustainable practices regarding tourism. We further conclude that the ECST has positive effects on biodiversity conservation, but that key actions in this respect have not been undertaken, such as the monitoring of bioindicators to value the effects of visitors' frequentation. The ECST was further effective because both the diagnosis and the Action Plan were accomplished using participatory techniques that ensured the participation of stakeholders, which was one of the objectives of the ECST. We have reviewed the situation of the actions planned under the five strategies analysed of the ECST until the 6th of November 2015 (see Table 6.2). From a total of 44 actions planned:

- 15 actions were finished;
- 15 actions were started and are still ongoing;
- 6 actions were initiated;
- 8 actions not undertaken.

Table 6.2 Classification of the actions of each strategy of the ECST analysed according to its degree of implementation

Situation of the actions	**Strategy 1**	**Strategy 2**	**Strategy 5**	**Strategy 6**	**Strategy 7**	**Total actions**
Actions finished	4	1	5	2	3	15
Actions started and still ongoing	4	4	4	2	1	15
Actions initiated	-	5	-	-	1	6
Actions not undertaken	-	3	3	-	2	8
Total actions per strategy	8	13	12	4	7	44

We have the following results when analysing each strategy separately:

Strategy 1. Cooperation and participation of different stakeholders of the territory

This strategy sought to integrate the participatory process of elaborating and monitoring the ECST in the already existing bodies of the natural park. The participatory bodies of the ECST are found effective because decisions taken in the Working Group and finally approved in the PF are binding. However, the communication between the Working Group and the PF should be enhanced, for instance, sending the minutes of the first body to people attending the Forum. This strategy further pursued to work in collaboration with companies related with tourism and to establish mechanisms for disseminating the ECST. It had eight actions, among which four finished and four started and are still ongoing. One of the actions was to join the Advisory Committee – an already existing participatory body, see section 5.3.2 for an explanation of the functioning of the Advisory Committee – and the PF because some aspects were shared implying a duplication of functions. However, during the workshop, some stakeholders were of the opinion that they should not be joined because each body had different competences; moreover, decisions by the PF were binding as opposed to those by the Advisory Committee not. A second action undertaken was to organise an annual workshop to make businesses more aware of the importance of the ECST. This was found to have positive impacts and reinforced the involvement of tourism sector in the ECST.

Strategy 2. Monitoring and evaluating the natural and cultural heritage.

This strategy was focused on inventorying the natural and cultural values of the natural park, monitoring the impacts of visitors on these values and promoting its conservation by distributing information among tourists about good practices. This strategy had 12 actions, among which one was finished, three started and are still ongoing, five had been started and three were not undertaken. Several inventories have been done by biologists of the natural park, such as of flora and emblematic trees, invertebrate fauna, vertebrate fauna focused on following and monitoring the wild boar (*Sus scrofa*) population. The latter is of paramount interest as there are many of them and they occasionally cause damages to farming crops. Park managers and employees considered that there is evidence that indicates that current conservation policies are functioning well. In 2012 a couple of Egyptian vultures (*Neophron percnopterus*) settled in the natural park and data collected about chiropters show they have increased. Two of the actions not undertaken were the identification and monitoring of bioindicators in order to value the effects of visitors' frequentation. In concrete, the species that were going to be monitored were bio-indicator species of the taxonomic groups of invertebrates, cryptogams, screws and vascular plants. This is particularly relevant because one of the challenges of the natural park in ecological terms is the impact that tourism might have in its flora and fauna.

Strategy 5. Heritage valorisation through creating tourism products.

This strategy promoted the creation of tourism products with added value as a result of the joint work between public and private sectors and the collaboration of local population, particularly in promoting new services for accommodation in rural areas. This strategy had 12 actions from which five were finished, four started and are still ongoing and three could not have been implemented. The main actions undertaken have been to participate in exhibitions and fam-trips, the creation of local tourist products based on the traditional way of producing wine and the enhancement of the cultural heritage of Talamanca village. This last action was particularly relevant because it included the restoration and conservation of a stretch of the river Llobregat and the valorisation of the cultural heritage of the *tines* – a traditional way of producing wine –. The actions not implemented, because of lacking budget, included the creation of a network of paths to promote hiking and a workshop to create new tourist products.

Strategy 6. Promotion of the education and knowledge of the natural park.

This strategy sought to analyse the training needs of the tourist agents of the territory and the existing offer of environmental education, and develop a joint training and educational plan. This strategy had four actions, from which two finished and two started and still ongoing. The actions undertaken were a census of the environmental education companies existing in the natural park and a participatory process with these companies to develop a common environmental education program. The actions that will continue are the elaboration and execution of a calendar of training sessions for the staff of the natural park and people from the private sectors, such as restaurants, farmers and environmental education companies, in order to provide quality information to visitors.

Strategy 7. Promotion and development of the local economy.
This strategy was focused on involving the population of the municipalities of the natural park in its advisory bodies, keeping informed the private tourism sector operating within the ECST and promoting local products and economic activities within the park. It included 7 actions; from which three of them were finished, one started and is still ongoing, one had been initiated and two were not developed. One of the actions of this strategy was to create a working group of the primary sector in order to analyse its situation, enhance coordination among farmers and park managers and ensure its development. However, this working group did not thrive since it only had one meeting. Another action related to the primary sector that was undertaken was the centralization of different projects for the preservation and production of local landraces. However, the involvement of projects developed by farmers was not promoted and finally, park employees were the ones who undertook the action through planting different local landraces in a plot inside the natural park. Another action related to farmers was the creation of a system for sharing agricultural machinery among them but it did not succeed. According to the representative of the farmers that attended the workshop, the enhancement of the primary sector, specially agriculture and livestock, in the natural park was not properly undertaken due to little interest from park managers, even though the working group for the primary sector had goodwill from the farmers initially. In fact, the farmers continued to be organized in the already existing "Association of farmers from the natural park" despite the unsuccessful attempt to promote the working group. There was a progressive decrease in organic farmers in the last decade in the area of the natural park. We consider that this decrease needs to be addressed because it implies a loss of local traditional knowledge and biocultural diversity. Furthermore, the lack of promotion of agriculture may be a problem because it has a central role in ensuring open areas that diminish the risk of fires in the area, which is one of the problems due to an increase of forest cover. Actually, one of the challenges of the natural park is to maintain certain diversity of ecosystems combining open spaces with forests. Some fields were opened through the cultivation of vineyard in certain areas of the natural park while in others livestock was increased.

One of the actions executed was the "Parc a Taula [Park on the table]" project, which was based on promoting contact between the primary sector, restaurants and the local population in order to strengthen collaboration. However, farmers claimed that this initiative was not effective for them because, at the end, it was focused too much on attending fairs which meant spending a lot of time and often they did not sell anything. They considered that the main aim of this initiative should be to promote the purchase of organic local products among restaurants and local inhabitants and to establish direct selling systems between farmers and their consumers that would shorten the distribution chain. Among the actions not executed is the enhancement of economic activities in villages through the development of the Plan for the future of the municipalities of Mura, Talamanca and Rocafort.

The second stage based on adhering companies to the ECST started in 2013 and also worked well. A total of 14 companies adhered to it. According to the attendants to the workshop, another aspect which decreases the effectiveness of the ECST is its high level of bureaucracy and complexity. This complexity generates a lack of understanding among participants at the PF when discussing certain topics.

6.4.2.2 Efficiency

We could not address this criterion because we do not have enough information to assess its cost-effectiveness.

6.4.2.3 Equity

We considered three types of equity: a) equity in terms of access, i.e. local people's ability to participate in the ECST; b) equity in terms of decision making, i.e. participants' perceived fairness in ECST decision-making procedures; c) equity in terms of outcome, i.e. focused on the impact and distribution of ECST outcomes.

In the case of equity in access, representation in the Working Group was unequal because some categories of stakeholders were absent and need to be represented in it, such as the conservationist sector and forestry organisations. Results from the attendance to the PF indicated there has been a significant change of the categories of stakeholders represented on it. Several trends can be identified (see Table 6.3). First of all, the presence of park managers, leisure sectors and tourism sector has increased significantly. Secondly, the scientific and the civic sector have diminished considerably their presence in the body. Thirdly, the agricultural, the conservationist and the forestry sectors have completely stopped attending the body. The reason given was that they felt the body was not effective to devote their scarce time to and that the body did not represent their interests. In the case of the agricultural sector, the representative explained that park managers did not support their proposals, like the agreement to set up the working group of the primary sector.

Table 6.3 Attendance to the Permanent Forum through time by different categories of stakeholders.

Different periods	**Local administration**	**Park managers**	**Park employees**	**Agricultural**	**Scientific**	**Civic**	**Conservationist**	**Leisure**	**Environmental education**	**Forestry**	**Tourism**
PF_0	22.7	4.5	6	4.5	7.6	12.1	3	6	15.1	3	15.1
PF_f	22.9	10.4	4.2	0	2.1	6.2	0	10.4	14.6	0	29.2
Tendency	+	++	-	--	--	--	--	++	-	--	++

Notes:
Results are expressed as percentages relative to the total number of different persons attending the meetings. When a person attended to more than one meeting of the same period of time, it was counted just once. We considered two different periods of time:
PF_0: attendance to the first meetings of the PF [66 different persons in 3 meetings].
PF_f: attendance to last meetings of the FP [48 different persons in 2 meetings].
The last row shows the tendency of changes in attendance by the different categories of stakeholders to the participatory body. ++ means considerable increase of the presence of the category in the body; + means increase; - means reduction; -- means considerable reduction.

In order to assess the equity in decision-making, we analysed the perceived influence and relevance of each category of stakeholders from a scale from zero to five, with zero denoting “no influence/relevance” and five “a lot of influence/relevance” considered by the attendants to the workshop. The categories that had more perceived influence in the PF were park managers (4.38) and environmental education sector (4). The ones having less influence were the forestry sector (2.38) and agricultural sector (2.75). The categories having more perceived relevance in the PF were the conservationist sector (4.11), park managers (4), tourism sector (4), leisure sector (3.89), environmental education (3.89) and agricultural sector (3.78). The ones having less perceived relevance were the park employees (3.11) and the civic sector (3.44). The fact that the most perceived relevant stakeholder categories (conservationist sector and agricultural sector) have stopped attending it shows that there is no equity in decision-making.

In the case of equity in outcome, the agricultural sector claimed that the promotion for the companies belonging to the primary sector was low. One of the farmers noticed that *“one of the approved proposals was to establish a working group of the farmers in order to set up collaboration, but park managers did not like this idea and hindered it.”* As a result, most farmers stopped participating because their interests were not taken into account. The outcomes are a bit unbalanced, favouring tourism and environmental education companies, while other sectors, such as primary sector, are not receiving enough attention.

6.4.2.4 Legitimacy

The ECST appeared legitimate to most stakeholders. Regarding its process of implementation, all stakeholders participating in the workshop except the persons belonging to the agricultural and conservationist sectors, considered the PF a legitimate body. The main arguments were that it promoted participatory dynamics (such as workshops and evaluative focus groups) and discussions among the participants and many people from the tourism and environmental education sectors attended it. However, there was no consensus on the reasons for promoting participatory techniques at the PF. Some stakeholders said that they were intended to collect different perspectives to establish the strategy of the ECST. Others pointed out that park managers merely organized participatory processes because they were mandatory to obtain the Charter certificate. Regarding the dynamics of the PF, stakeholders in average thought that this body was informative with a punctuation of 3.75 on a scale from zero to five. In addition, it scored on interactive (3.57), advisory (3) and decisive (2.57). These results show there were joint discussions among participants during the meetings of the participatory body in order to reach agreements.

During the workshop, the person from the agricultural sector considered that many people attended this body which made it highly time consuming and little constructive. Some argued that “most of people going to the meetings of the ECST have direct interests, both professional ones such as park workers, managers or scientists; or economic ones like small-scale companies related with tourism”. Regarding the legitimacy of its content, some stakeholders pointed out that it was overly focused on tourism issues and other necessary topics such as agriculture were not receiving enough attention.

6.4.2.5 Complementarity of policies on one or multiple criteria

The official documents of the ECST already identified several complementarities among actions of the ECST. For instance, they described that one action of strategy 1 focused on organising an annual workshop to make businesses more aware of the importance of the ECST reinforced stage 2 of the ECST, which focused on certifying businesses with the ECST. Another complementarity described was that one action of strategy 6 based on elaborating a census of the environmental education companies existing in the natural park complemented with the action of the same strategy 6 focused on developing an environmental education program through a participatory process with these companies.

We found that it was further interesting to identify complementarities of the actions of the ECST with other policy instruments implemented in the natural park. For instance, the strategy 2, focused on monitoring and evaluating the natural and cultural heritage, is reinforced by the Monitoring Plan of ecological parameters and with actions done to restore certain habitats, such as pine forests, ponds and fountains, eliminate invasive species like the tree of heaven (*Ailanthus altíssima*) or actions to conserve certain species of reptiles and amphibians. It is also complementary with the agreement made with climbing entities to regulate this sport in order to protect chiropters and other species living in caves. Some actions of Strategy 2 are focused on evaluating the impacts of visitors on flora and fauna. The mentioned agreement is an example of an instrument designed after it was assessed that climbing in certain locations and periods of time had negative impacts on certain species living in caves.

Another policy instrument that complements strategy 2 is the Conservation Plan. This is still being drafted and aims to establish conservation priorities in the natural park based on previous inventories and studies regarding different species and habitats. Therefore, actions of strategy 2 focused on inventorying and monitoring species can provide information to establish such conservation priorities.

There is a program of subsidies from the Diputació de Barcelona to service companies operating in the area of the natural park which subsidize the improvements needed by these companies in order to be certified by the ECST. These subsidies reinforce strategies 5 about heritage valorisation through creating tourism products, strategy 6 focused on promotion of the education and knowledge of the natural park and strategy 7 about promotion and development of the local economy.

Strategies 5 and 6 are complementary with the Program “Passejades els diumenges” [Sunday Tours] which promotes companies of environmental education and restaurants organizing tours along the natural park. Other cultural and gastronomic activities were planned focused on organizing a trip together with a lunch in a restaurant of the area but they did not work because not many people demanded these types of activities.

The actions established in strategy 7 are complementary with the subsidies to agriculture of the Diputació de Barcelona and the one from the Generalitat de Catalunya. The subsidy from the Diputació is higher if the agriculture is organic and following good practices as mentioned by the director of the park, although such practices are not specified in any document.

6.4.2.6 Overlap of, or conflict between, policies in terms of a particular criterion

There are no concrete actions of the ECST that overlap or conflict between each other. However, there are actions that were not performed that might minimize the effectiveness of other actions. For instance, the action of evaluating the impacts of tourism on certain species has not been done and could have generated necessary inputs for the actions focused on promoting sustainable tourism. We have further identified potential overlap or conflict of actions of the ECST with instruments implemented in the natural park. We have pointed out that the permission of hunting in order to control the population of wild boars might be in conflict with the actions designed to enhance tourism uses of the area and other uses such as hiking. The presence of hunters in the area associated with the sound of gunfires, might be a disincentive for potential visitors to come to the area during hunting days. Hunting days are announced on the website of the natural park and through posters in the main roads and paths of the natural park.

6.4.2.7 Monitoring and control

The implementation of actions is evaluated in the meetings of both the Working Group and the PF. There is a process of evaluating the whole ECST every five years. In order to obtain again the certification, certain aspects are checked by Europarc Federation in the evaluation process, such as: 1) the continuity and increasing progress in its work, the existence of a PF or similar structure for the development of sustainable tourism in the area, the spread of information about the Forum's activities (for example events, decisions, etc.) and evidence proving that all relevant stakeholders are involved; 2) the existence of positive and reinforced cooperation with local businesses; 3) clear progress in implementing the action plan that was presented in the original application; 4) the elaboration of a strategy and a plan of action for the next five years publicly available. There are actions that include specifically a monitoring process, such as the tracking and monitoring of wild boar (*Sus scrofa*) populations.

6.4.2.8 Creation of incentives for agents to alter their behaviour

The incentives for protected areas to adopt the ECST are the economic, social and environmental advantages of well-managed, sustainable tourism. The Charter also gives participating protected areas a basis for strengthening relationships with local tourism stakeholders and the wider tourism industry in order to promote the adoption of sustainable practices and the opportunity to influence tourism development in the area. Another incentives to be certified with the ECST is to have a higher profile in the European arena as an area devoted to sustainable tourism; promote the awareness-raising among visitors and local and national media about sustainable tourism; the opportunity to work with and learn from other European Charter areas in the Charter network; to have helpful internal and external assessment that may lead to new ideas and improvements in management and finally to have greater credibility amongst potential funding partners. The fact that 14 companies adhered to the ECST reveals that they have enough incentives to join the policy instrument. There are a wide range of them such as economic ones like more promotion of the company and satisfaction of developing better environmental practices.

Participatory bodies, such as the PF, have changed the attitude of certain stakeholders (mainly members of hiking groups and conservationists) who were against policies that promoted the public use of the area. They thought that public use was in conflict with conservation of the natural park. Communication through participatory bodies has diminished conflicts between park managers and these stakeholders although some conservationists have manifested during the workshop that there were a lack of trust in participatory bodies, reason why they stopped attending them.

6.4.2.9 Consistency with institutional-cultural context

As a policy instrument ECST is consistent with the cultural and institutional context. The well-functioning of its core, namely the voluntary participation and commitment of stakeholders with the conditions of the ECST, requires a predisposition to participate. Stakeholder participation can be facilitated if a social network exists linked to the protected area. The establishment of collaborative governance and management initiatives like the ECST need the existence of a previous social network that guarantees access to information by stakeholders and the building of relations of trust between people. If not, it is difficult that stakeholders want to commit with the policy instrument. In the case of the natural park of Sant Llorenç, there is a network composed of 238 stakeholders belonging to different categories of stakeholders, as is shown by the results of the Social Network Analysis done in chapter 4. Regarding the institutional context, the implementation of the governance structure of the ECST - the Working Group and the PF - was feasible due to the previous existence of similar participatory bodies in the institutional context of the natural park such as the Advisory Committee. There was an adaptation of the functions of these new participatory bodies with the previous existent ones in order to avoid duplication. Actually, the Advisory Committee and the PF were joined in the same meeting.

6.4.3 Identification and characterization of rebound effects of the ECST

Here we discuss potential rebound effects of the ECST. The examples of rebound described below are merely suggestive as we do not have scientific data yet to support them. However, we believe that a preliminary proposal of potential rebound effects can provide valuable information to avoid future negative indirect effects.

1. Biodiversity rebound I: There is a high amount of visitors concentrated in the central area of the natural park, which is the most fragile in terms of habitats and species. Some actions, such as strategy 7 of the ECST, promote the redirection of visitors to northern areas of the natural park, which can lead to the increase of ecological impacts due to tourism in areas that were previously more isolated.

2. Biodiversity rebound II: One of the actions not already done of strategy 2 is monitoring the population of flora and fauna species as bio-indicators to assess the impacts of tourist frequentation, especially trampling. In concrete, the species that are going to be monitored are bio-indicator species of the taxonomic groups of invertebrates, cryptogams, screws and vascular plants. We consider that this is a good starting point although it has not been already developed. However, it may lead to focus the attention on the effects of tourism only in these taxonomic groups and not considering the impacts to other taxonomic groups or other type of biodiversity such as functional diversity. For instance, high amount of visitors can affect functions related with soil formation leading to soil compaction.

3. Ecological rebound: One of the actions of strategy 2 is based on monitoring wild boar populations. Depending on the results of this monitoring study, the future solutions proposed to deal with the high number of wild boar may generate changes in the wild boar population. For instance, the proposal of an increase in hunting would lead to less population of wild boars, or the limitation of hunting for protectionist reasons would lead to an increase of the population. These changes in the quantity of wild boars may affect the trophic chain of this animal. This can give rise to a loss of equilibrium between different species in the ecosystem because of food scarcity or predator pressure. In this sense, we consider that this monitoring of wild boar populations should include also monitoring of the impacts generated by this animal on flora, fauna and other biophysical components, this is, its impacts in a broad ecological terms.

4. Service rebound: The permission of hunting in order to control wild boar population may be in conflict with the tourism uses in the area. This is already explained in the criterion addressing overlap of, or conflict between, policies. Another potential service rebound is the one generated by the agreement made to regulate climbing in order to protect chiropters and other species living in caves. It is in conflict with climbing activities, because the access to certain caves during concrete periods of time is forbidden.

5. Environmental rebound: The promotion of sustainable tourism in protected areas as a way to ensure that visitors reduce their impacts on biodiversity may itself already increase general pressure on the ecosystem by promoting high number of visitors in protected areas that will have negative impacts in terms of conservation of biodiversity. This may mean a shift in priorities of protected areas, leading to value them in terms of tourism and generation of income instead of focusing on conservation as the first aim. The increase of the frequentation of visitors in the area due to the promotion of sustainable tourism may lead to the increase of domesticated animals which can increase the disease risks among wild animals due to the contact with the former ones. Another rebound of the promotion of sustainable tourism is the pollution generated by emissions and noise from cars, together with waste and noise during recreational trips.

6.4.4 Comparison of different policy instruments of European case studies

This section summarizes the results of the evaluation criteria of four other instruments dealing with biodiversity conservation and the provision of ecosystem services implemented in Europe, together with the results of the ECST implemented in the natural park of Sant Llorenç. The evaluation of the instruments was undertaken by partners of the European project CONNECT and the case studies were selected in order to cover different European representative landscapes, namely the North-West European delta landscape, the French Alpine mountain landscape, the Central European agricultural and forest mosaic landscapes, the Scandinavian forest landscape and the Mediterranean landscape. The case studies are 1) a comparison between dikes and shores along or off the coast to control flood probabilities and maintain wildlife in Ijsselmeer, a large fresh water reserve in the Netherlands; 2) an analysis of 10 instruments dealing with agriculture, tourism and biodiversity conservation in the French Alps, 3) an evaluation of an incentive-based agri-environmental scheme for afforestation in the state of Saxony, Germany; 4) a description of 7 groups of policies important for biodiversity conservation in Swedish forests; 5) an analysis of the European Charter for Sustainable Tourism implemented in the natural park of Sant Llorenç, in Catalonia (Spain). Table 6.4 characterizes the results of the different criteria implemented to these policy instruments.

Table 6.4 Implementation of the criteria to different policy instruments from European case studies[18]

Criterion	**Case study Netherlands (NL)** **Comparison between dikes and shores along or off the coast to control flood probabilities and maintain wildlife** [1]	**Case study France (FR)** **10 instruments**[2] **dealing with agriculture, tourism and biodiversity conservation**	**Case study Germany (DE)** **Incentive-based agri-environmental scheme for afforestation**	**Case study Sweden (SE)** **Description of 7 groups of policies**[3] **important for biodiversity conservation in Swedish forests**	**Case study Catalonia (CAT)** **European Charter for Sustainable Tourism implemented in a protected area**
Effectiveness	a) Relative performance of dikes and shores in reducing flood probabilities. Dikes need to be sufficient high and shores sufficient large. b) Maintaining a certain quality of nature and its function as a wildlife habitat is uncertain for both instruments. Dikes imply buying land elsewhere to compensate so conditions and consequences for wildlife and bird populations will be different. Creating shores seems more effective than buying up land elsewhere.	(+) PNAL, PHAE2, IG: demonstrated support for mountain farming that positively impacts environmental quality. (-) AeA: restricted scale and no direct environmental objective. PDR: little actual environmental gains and stakeholder collaboration. UTN: being a derogation from a more conservative strategy (Mountain Law).	Low effectiveness: The goal is to increase the forest cover to at least 30 percent at a decent pace. The forest cover shall increase to about 28.8 percent by the year 2020 and keep growing by 0.4 percent per decade in the following thirty years. Between 2007 and 2009 a total of 57.96 ha of agricultural land was afforested which equates to an increase of forest cover of only 0.01% in Saxony.	(+) Forest Act: Specific regulations (size of the logging site, fertilization, drainage and amount of trees left on the logging site) that have a positive effect on biodiversity by making it possible for species dependent on certain substrate and structures to survive. (+) Economic instruments (subsidy for maintenance of broadleaved deciduous forests and maintenance funds for PAs): preservation of function and structures in the forest	(+) Most actions undertaken. The first stage consisting in elaborating the diagnosis and the Action Plan together with stakeholders through participatory techniques worked well. The implementation of the Action Plan further worked well and it has been effective in addressing certain ecosystem services and biodiversity conservation. The second stage based on adhering companies to the ECST also worked well (14 companies adhered). Decisions taken in

[18] For more concrete information about the European case studies see the report of the European project CONNECT: Maestre-Andrés S., Crouzat E., Koetse M.J., Nordén A., Völker M. 2016. A synthesis report of case studies results on the analysis of policy instruments and governance structures for biodiversity conservation.

				and for protecting species dependent on certain habitats. (-) Protected areas: variation in PAs is below sufficient levels with clear differences in the landscape and habitat types represented. Not even half of the species in the country are represented in the formally PAs. PAs are too small and too scattered for the policy targets to be reached.	the Working Group and finally approved in the PF are binding, which ensures the effectiveness. (-) The ECST implies high level of bureaucracy and it is complex.
Efficiency	Data missing. A potential problem is that there is limited or no monitoring of the effects of these shores on flood probabilities and maintaining wildlife habitat.	(++) ENS, PAEN: offer a perennial environmental protection on areas undergoing human pressure. (+) AeA, IG: very limited budget. (-) PDR, PNAL, PHAE2, PTCA: they rely on substantial budgets. PHAE2 supports already existing practices thereby not creating additional environmental gain (lack of additionality).	Not addressed due to data limitations.	Not addressed due to data limitations.	Not addressed due to data limitations.
Equity	Dikes have implications of the potential change of the location of nature and wildlife habitat. Increasing dike height will affect (positively or	(++) PDR, PHAE2, PNAL, AeA: they compensate for additional constraints farmers face in mountain areas.	(+) The scheme is considered fair because i) everybody interested can apply, ii) it is voluntary and iii) farmers with no		3 dimensions of equity analysed: a) equity in access, b) equity in decision-making and c) equity in outcome. a) (-) Equity in access: there

	negatively) people who live directly next to it. Creating shores will benefit mainly the people who use them for recreation purposes. Both policies will imply larger supply of fresh water to meet the expected increase of demand. Who assumes the costs of the policies (levied through local taxes or through nation-wide income taxes) is determinant for equity implications.	(++) PAEN, ENS: they restrict some land uses (urbanization) and thereby exclude some stakeholders (deny private interests) but tend to reinforce global equity by keeping these areas publically accessible and in good environmental condition. (+) the rest of instruments (-)UTN: high costs are restrictive for small municipalities. IG: promotes differentiation of agricultural products and does not treat equally all farmers.	forestry experience have the right to apply for subsidies. (-) unfair in inter-generational sense because the benefits of afforestation accrue far in the future while its costs need to be covered today. No decision-power to plant forest in rented land, negotiations with landowner are regarded to be difficult. Large farms have an advantage over smaller farms with regards to their suitability for afforestation because they can still use sufficient land for agricultural purposes.		is unequal representation in the PF of the different categories of stakeholders. Some categories (farmers, conservationists and forestry sector) have completely stopped attending the body. The reasons are the perception that the body is not effective, it does not represent their interests and stakeholders' lack of time. b) (-) Equity in decision-making: there is unequal relevance and influence among categories of stakeholders. The most perceived relevant stakeholder categories (conservationist sector and agricultural sector) have stopped attending the PF. c) (-) Equity in outcome: Low promotion for the companies belonging to the primary sector, it involves an extra effort for them. One of the approved proposals was to establish a working group of the farmers in order to set up collaboration, but park managers did not like this idea and hindered it.
Legitimacy	Substantial increases in	(++) SRCE: participative	Approach: fairness of the		(+) The instrument appears

	welfare may be obtained by using shores. WTP estimates are substantial for the creation of shores, for reducing flood probabilities and for (partially) conserving the Ijsselmeer bird population.	process. (+) the rest (-) UTN: impartiality was questioned by some stakeholders due to lack of information on the actual elements of justification for positive/negative derogatory decisions. Current debates focus on implementation (specific location, management practices) or on budget allocation.	process of decision-making. Farmers were not involved in the design of the scheme, which suggests a lack of legitimacy. However, farmers also stated that they do not mind not having been involved.		legitimate to most stakeholders. Legitimacy of the process: (+) all stakeholders, except the persons belonging to the agricultural and conservationist sectors, considered the PF legitimate. The PF promotes participatory techniques. (-) Legitimacy of content: it is overly focused on tourism issues and other necessary topics such as agriculture are not receiving enough attention.
Complementarity of policies	The creation of shores may be complemented by an increase in dike height, for example because shores alone are not sufficiently effective in reaching the required goal, or because a combination of the two measures is more cost effective.	All instruments presented many complementarities with instruments of diverse natures and related to various scales.	Between the scheme and afforestation projects implemented through offset measures (compensations for environmental damages caused by developments). Farmers are not restricted in their participation in other environmental measures.	Subsidy for maintenance of broadleaved deciduous forests and maintenance funds for protected areas are complementary to the legal regulation.	Strategy 1 focused on organising an annual workshop to make business more aware of the importance of the ECST reinforced stage 2, which deals with certifying businesses with the ECST. Strategy line 5 “Promotion and development of the local economy” is complementary with the line of subsidies to organic agriculture of the Diputació de Barcelona and the one from the Generalitat de Catalunya. It is also complementary to the “Parc a Taula” project.

					The strategy line 2 "Monitoring and evaluating the natural and cultural heritage" is complementary with the Monitoring Plan of ecological parameters, with actions undertaken to restore certain habitats, to actions undertaken to eliminate invasive species and actions to conserve certain species of reptiles and amphibians. The Conservation Plan is another complementary policy. It is also complementary with the agreement made with climbing entities to regulate climbing to protect chiropters and other species living in caves.
Overlap and/or conflict between policies	When the creation of shores is sufficient to reach a predefined flood risk level, increasing the height of dikes does not ad anything (although it may reduce flood probabilities even further).	Positives redundancies: PDR – PHAE2: with the Interregional Convention for the Alpine Massif (CIMA) and the 'Agriculture' protocol of the Alpine Convention. SRCE: with zoning for protection at lower scale. ENS: with other small-scale protected areas, including with PAEN. Negative overlaps:	The visibility of the scheme is compromised by numerous alternative funding opportunities for farmers (support for biofuel and short rotation forestry).		No overlap between actions of the ECST. Potential conflict was identified between actions of the ECST and other instruments implemented in the natural park. The permission of hunting in order to control its population might be in conflict with the instruments designed to enhance tourism uses of the area.

		PAEN: with specific protective status for agricultural areas undergoing artificialisation pressures (ZAP). UTN: with instruments promoting conservation objectives. PTCA: with local preferences for ski resort development. PNAL: with tourism-related instruments like AeA and instruments supporting extensive pastoralism.			
Monitoring and control	Both instruments need to monitor their quality occasionally. Dikes have large up-front but low maintenance costs while shores can be created at relatively low costs (although this depends strongly on the depth of the water) but require more maintenance.	(-) "under-optimal" for PTCA, PHAE2 and UTN. In the case of UTN procedures of control exist but some stakeholders fear that they are sometimes by-passed. They criticise the lack of transparency of its environmental assessments.	(+) Administrative and on-site control measures are conducted by state and external agencies. Information on associated costs is not available.	(-) Voluntary agreements: weaker than formal protections since there are no monitoring possibilities. (+) Information measures (red list that estimates the risk of extinction of certain species): it is based on an expert committee's evaluation, environmental monitoring and other data.	(+) The implementation of actions is periodically evaluated in the meetings of both the Working Group and the PF. There are actions that include specifically a monitoring process, such as the tracking and monitoring of wild boar (*Sus scrofa)* populations. There is a process of evaluating the whole ECST every five years.
Creation of incentives	Not that they can see.	Not mentioned	(-) There are not enough incentives to participate in the program, therefore participation rate in the scheme is low. Subsidy levels offered by the agri-	(+) voluntary agreements: incentives for set-asides are driven by certifications (FSC and/or PEFC). Increase in price on production that can	(+) Protected areas which meet the requirements of the Charter will benefit from the economic, social and environmental advantages of well-managed, sustainable

			environmental scheme are well below the opportunity costs of afforestation which means that farmers would need to accept a reduction of their income when participating in the scheme. There are also institutional constraints such as the complicated application procedure and the limited number of application deadlines per year (only one deadline per year).	financially compensate the forest owner for setting aside forest. (+) conservation agreements imply monetary compensation from the government. (-) Tax regulation: the incentives created through the tax system are important for forest owner when deciding about forest measures. Most of them increase incentives for harvesting and less for conservation measures.	tourism. 14 companies adhered to the ECST, which reveals that they have enough incentives to join the policy instrument. There are a wide range of them such as economic ones like more promotion of the company and satisfaction of developing better environmental practices.
Consistency with institutional-cultural context	Creation of dikes is the traditional way of protection against floods. Shores are less traditional which might be considered to be a less preferred option. The choice experiment reveals that people are positive about increasing the height of dikes and are also positive about the creation of shores (but in the experiment this choice attribute was not directly linked to flood probability).	All instruments are consistent.	Consistent. Financial support programs have a long tradition in agricultural and conservation policies. Recent shifting of the responsibility for the scheme from forest authorities to agricultural authorities has led to a loss of interest and competences in advertising the scheme, and farmers can apply to a number of other support schemes closer to their core interests of actually farming on their	Even though production and environmental goals should be regarded as equally important according to the Swedish Forest Act, forest owners, public forestry officials and employees at industrial forestry companies and forest owners associations prefer management practices that promote production rather than biodiversity protection. The fact that this bias in preferences is particularly evident for private sector employees	The policy instrument of the ECST is consistent with the institutional and cultural context. The well-functioning of its core, namely the voluntary participation and commitment of stakeholders with the conditions of the ECST, requires a predisposition to participate. Stakeholder participation can be facilitated if a social network exists linked to the protected area. There is a network composed of 238 stakeholders belonging to different categories of

	agricultural lands.	might lead to a continuous focus on production rather than biodiversity protection as forest owners currently have more contact with private companies and forest owners' associations than they do with public officials.	stakeholders that shows the potential for the establishment of collaborative governance and management initiatives such as the ECST.

Notes:

[1]No actual policies implemented, just possible policies to deal with future changes. These results are from a choice experiment on consumer preferences for potential future changes in the Ijsselmeer area.

[2] Authorisations for New Tourism Facilities (UTN), Regional Scheme for Ecological Coherence (SRCE), Tourism Protocol of the Alpine Convention (PTCA), Wolf National Action Plan (PNAL), Regional Plan for Rural Development (PDR), Grass premium from the CAP - second pillar (PHAE2), Geographical Indications for Agricultural Products (IG), Pilot project for Tourism Diversification in pastoral activities (AeA), Protective Zoning for Natural and Agricultural Areas (PAEN), and Protected Sensitive Natural Areas (ENS).

[3]The seven policy instruments are the Forest Act, specific economic policy instruments, protected areas and species, voluntary agreements, tax regulations, information measures, and hunting regulations.

The performance of some criteria was evaluated following four possible results: (++) very good, (+) good, (-) bad, (--) very bad. Some criteria were not evaluated following these four options because their performance needed to be more nuanced and explained or because the performance of the criteria was discussed in a hypothetic scenario (the Netherlands case study).

The comparison of the various policy instruments in the different countries is complicated, as there are many contextual differences. Hence, we have summarized the main results and provide practical guidelines and lessons for better design of biodiversity policy. In the Dutch case study, creating shores along the coast as a policy measure to deal with the increasing water levels appears to be a promising alternative to the more traditional measures like increasing the height of dikes and buying up land elsewhere. However, the effects of the different policies (including the no policy option) on ecological indicators, biodiversity and on ecosystem services, have not been assessed so far. The results of the study show that people legitimate more the creation of shores because they reduce flood probabilities while conserving bird population.

Regarding the French case, economic instruments were assessed as inefficient mainly because they imply a substantial budget allocated to support alpine farmers but they had a limited environmental effectiveness, hence referring to cost-effectiveness. Indeed, the PNAL budget of more than 10 million euros per year is to be used to promote a better cohabitation between French alpine pastoralism and 'only' one specific species. Moreover, most PDR measures have been assessed as costly compared to their actual environmental effectiveness. And PHAE2 presented a lack of additionality, i.e. the incentive was used to promote practices that would have been adopted regardless of the instrument's existence. However, these three instruments were found to favour equity. The SRCE was the instrument most legitimate because it followed a participatory process. UTN was the instrument more questioned in terms of environmental effectiveness, equity, legitimacy and its lack of transparency in its environmental assessments. Overall, the French policy mix was characterized by many complementarities and effectiveness, although presenting several overlaps between instruments. This is achieved through mechanisms like voluntary agreement complementing direct regulation such as the Pastoraloup – an eco-volunteering program aiming at helping farmers and shepherds to protect their herds – being complementary to the national PNAL providing support to farmers in areas where wolves are reported.

In the German case, the agri-environmental scheme was considered of low effectiveness. It was considered to favour large farms and farmers having rented land found difficulties to join the scheme, which has implications in terms of equity. The scheme was considered legitimate but there was a low participation rate. The reasons are that the subsidies are below the opportunity costs of afforestation, the application procedure is complicated and there is a limited number of application deadlines per year. Some solutions to the problems highlighted could be to hire rangers that can give advice to farmers regarding the implementation and management of the afforestation scheme. Farmers would thereby be released from some of the practical burden of afforestation. Involving rangers might also increase the efficiency of planting and managing the forest. Finally, changes to the administration of the afforestation scheme could further motivate enrolment. Farmers would like to have the opportunity to submit applications at any time as opposed to having a strict once-in-a-year application deadline. However, this would substantially increase transaction costs on the side of the authorities handling the scheme. Furthermore, farmers prefer a reduction in bureaucracy as well as better information and support during the application process.

Regarding the case study of Sweden, protected areas were considered ineffective because not even half of the species in the country were represented in them and were too small and too scattered for ensuring effective biodiversity conservation. The Forest Act and economic instruments were considered effective because they contributed to biodiversity and ecosystem services through preservation of functions and structures in the forest and for protecting species dependent on certain habitats. Its results showed that although public officials have higher preferences for biodiversity than their private sector counterparts, they also tend to prefer management practices that promote production. This conflict may have negative consequences for biodiversity protection, as public officials work directly with regulations and guidelines to impact how forest management is carried out in practice. This finding suggests that the Swedish Forest Act may need to be complemented by clearer enforcement tools in order to better reflect the prioritization of environmental goals alongside production targets.

In the case study of Catalonia, the ECST was considered effective. However, it was evaluated negatively regarding the three dimensions of equity identified. The instrument appeared legitimate to most stakeholders but its content was overly focused on tourism issues and other necessary topics such as agriculture were not receiving enough attention.

Several recommendations can be made considering the results. Stakeholder participation seems to be relevant in three out of the five case studies analysed. In the case of France, the most legitimate instrument was SRCE because it followed a participatory process. In the Catalan case, stakeholder participation had a central role in the implementation of the ECST. It increased its legitimacy and was necessary to promote the adherence of tourism companies to the ECST. In the case of Germany, the participation of farmers in designing the agri-environmental scheme could had resolved the low participation rate of farmers in the scheme by identifying the problems of it and proposing solutions. Actually, they provided interesting ideas to improve the scheme during the research. Our results show that the combination of policy instruments improves the performance of individual instruments by generating synergies among them. This is particularly relevant in the French, German and Catalan case studies were a more systematic analysis of different instruments was done. Therefore, it is important to design instruments from a perspective of a policy mix (Ring and Schröter-Schlaack, 2011), this is to consider the interaction of different instruments in order to improve their performance in terms of biodiversity conservation and ecosystem services provision. Moreover, it permits the identification of overlaps and conflicts between policies. Our findings found that few policies incorporate monitoring and control measures. Establishing these measures is considered necessary to reinforce the effectiveness of instruments and further assessing their implications in terms of efficiency, equity and legitimacy.

6.4.5 Characterization of rebound effects from different case studies

The following table (Table 6.5) characterizes the different rebound effects of each European case study:

Table 6.5 Characterization of rebound effects from European case studies.

Rebounds	Case study NL	Case study FR	Case study DE	Case study SE	Case study CAT
I	Buying up land elsewhere: nature and bird populations in and around the Ijsselmeer would decrease, while nature and biodiversity would increase at the locations where land is bought (it depends on the type of land and its management).	IG, PDR, PHAE2: a differentiated management focusing agricultural measures on specific, constrained and disadvantaged areas could lead to lower environmental standards for other areas such as valleys of lower cultural value or as lower rural areas. SRCE, PAEN, ENS: the protection of specific areas from land planning opportunities could increase land pressure on remaining areas that are also potentially of interest for biodiversity. UTN: new facilities for artificial snow withdraw water volumes and alter the annual water cycle. This disequilibrium can affect downstream biodiversity.	No rebound	No rebound	Strategy 7 of the ECST promotes the redirection of visitors to northern areas of the natural park in order to decrease the amount of visitors in the central area – the most fragile in terms of habitats and species. This action can lead to the increase of ecological impacts due to tourism in areas that were previously more isolated.
II	In buying up land elsewhere and creating shores, existing natural areas are replaced with new ones at different locations and with different characteristics. This may lead to differential effects on biodiversity, but an	PNAL: the concentration of herds in secured areas increases trampling and overgrazing, which alter biodiversity and favour more generalist species. SRCE: it could favour species with strong dispersal abilities that	While on average the predicted number of species per area was increased by increasing forest cover there were also areas for which the models predicted losing species richness due to	There are clear differences in the landscape and habitat types represented in PAs. For example, alpine landscape constitutes almost 70 % of the total protected areas while forests constitute	Strategy 2 is going to monitor certain species as bio-indicator to assess the impacts of tourist frequentation – it has not been already developed. These species are of the taxonomic groups of

	assessment of direction and magnitude is very difficult. Shores: it may maintain bird population but biodiversity may be affected because new wildlife habitats may stimulate different species than before. This may also affect fish population, especially when the number of fish-eating birds decreases. It depends on the flora used for creating the shores.	would benefit from the green and blue corridors to colonise new ecosystems. To date, this remains a hypothesis mentioned by the stakeholders and a current debate in the scientific literature.	forest cover change. Policies for afforestation may enhance forest biodiversity (depending on type of afforestation), but may be in conflict with grassland biodiversity, as HNV grasslands with low economic returns may be the first lands to change into forest. Forest cover change is predicted to lead to improvements in water quality. This is highly likely to affect freshwater biodiversity in the river system downstream of the afforested regions. As participation rates in the scheme were very low, these effects are small.	around 1%. For habitat types, less than 2% of the coniferous forest in the country is represented in protected areas, with a bit larger shares for the country's mixed forests (4.3%) and deciduous forests (13.9%). Probably prioritizing one type of habitat has led to the underrepresentation of other types of habitat.	invertebrates, cryptogams, screws and vascular plants. However, it may lead to focus the attention on the effects of tourism only in these taxonomic groups and not considering the impacts to other taxonomic groups or other type of biodiversity such as functional diversity. For instance, high amount of visitors can affect functions related with soil formation leading to soil compaction.
III	Both policies affect the relationship between fish populations and bird populations. Buying up land elsewhere does not mitigate decreases in bird population in the area. This depends on the location of buying land. If land is being created within the Ijsselmeer area, effects may not be very different between the two policies.	UTN: one direct effect is to artificialize ecosystems, which is negative for ecological functioning in general. IG: lack of coherence in the supply chain for products could negatively affect ecosystems. SRCE: it is uncertain the impacts of an increased connectivity on ecosystem functions as a	Increasing forest cover led to a significant reduction of nitrate loads in all parts of the basin.	Hunting regulation: for species that have a favourable conservation status, hunting is permitted during certain seasons and for certain circumstances. The impact of hunting on biodiversity is through the direct impact on the conservation of the animal being hunted, as well as on other species in the same	High variation in wild boar population (due to an increase in hunting or a limitation of hunting for protectionist reasons) may affect the trophic chain of this animal. This can give rise to a loss of equilibrium between different species in the ecosystem because of food scarcity or predator pressure.

		result of colonisation by invasive species. ENS: public over-use would be negative for ecological functioning (by over trampling).		food chain. Hunting one non-endangered species can give rise to a loss of equilibrium between different species in the ecosystem, affecting endangered species.	
IV	Changes in fish populations may directly or indirectly affect revenues for professional fishing. Not only fresh water supply increases because water levels increase, but also its quality may be affected by changes in biodiversity (species that filter the water may decrease in number). Shores may have a large impact on recreation potential along the coast. Area used for the creation of shores cannot be used anymore for sailing and surfing. If these shores become designated biodiversity reserves, sailing and surfing possibilities may decrease even further. It depends on how much the coast is currently being used for certain activities.	PNAL, PDR, PHAE2: extensive agricultural practices including environmental constraints in management usually decrease provisioning services. This situation echoes with the fact that agriculture itself decreases a number of regulation services provided by the forests which would otherwise grow at altitudes up to 2100 – 2400 m. PNAL: wolf return and adapted agricultural practises conflict with leisure hunting and recreation activities in higher altitude areas and also tend to impact landscape aesthetic quality. UTN, ENS, SRCE: we were not able to determine the dominant trade-off among ES categories because they depend on local	Afforestation of agricultural land means a decrease in provisioning services related to agricultural production. Landscape aesthetics might be positively or negatively affected, which is a matter of taste.	No rebound	The permission of hunting wild boar in order to control its population may be in conflict with tourism uses in the area. The agreement made with climbing entities to regulate climbing in order to protect chiropters and other species living in caves is in conflict with climbing activities, because the access to certain caves during concrete periods of time is forbidden.

		management modalities.			
V	Buying up land elsewhere may have negative consequences for water quality, while creating shores may not have these adverse effects.	UTN: it could increase the number of visitors in the Alps, thereby inducing an increase of greenhouse gas emissions, CO_2-intensive energy consumption and water pollution, etc. PHAE2: the decrease in food yields induced could be compensated by imports of forage that would induce spatial environmental rebounds and greenhouse gas emissions.	In landscapes with such low forest cover no negative impacts are expected.	No rebound	The promotion of sustainable tourism in protected areas as a way to ensure that visitors reduce their impacts on biodiversity may indirectly lead to enhance tourism, thus promoting high number of visitors in protected areas that will have negative impacts in terms of conservation of biodiversity.

6.5 Conclusions

The European Charter for Sustainable Tourism implemented in the Natural Park of Sant Llorenç del Munt (Catalonia, Spain) has been judged as an effective policy instrument to engage different types of stakeholders in designing and implementing strategies and policies to protect ecosystem services and biodiversity. However, it has been evaluated negatively regarding the three dimensions of equity identified. For instance, equity in terms of access to the ECST is not ensured as agricultural, conservationist and forestry organizations have stopped attending the PF, were decisions regarding the ECST are taken. The instrument has appeared legitimate to most stakeholders – except the ones that stopped attending the PF – but its content is overly focused on tourism issues leading to paying less attention to topics such as agriculture. In this sense, actions developed to support the primary sector have not been properly undertaken due to little interest from the park management, even though initially it enjoyed the sympathy and support from the farmers. The increased risk of fires in the area due to an increase of forest cover reveals the importance of the role of maintaining farming so as to ensure open areas.

There is good complementarity among the different strategies and actions of the Charter and with other instruments implemented in the protected area. These is little overlap or conflict among actions of the ECST and among other policies of the protected area. The functioning of the PF needs to improve in order to ensure the attendance of all stakeholders, a necessary condition to advance to effective collaborative governance initiatives.

Effectiveness of instruments as mentioned above is more complicated than often perceived. Different potentially unwanted and avoidable indirect effects – rebound effects – regarding the ECST have been identified as potentially reducing the effectiveness of biodiversity policies.

A comparison with other policy instruments implemented in European countries has been done. However, due to disparities in addressing different objectives and the different context of implementation, a consistent comparison has been difficult to achieve. Nevertheless, the main results of each instrument have been summarized and different rebound effects of each instrument have been identified in order to provide practical guidelines and lessons for better design of biodiversity policy.

Considering the results of comparing instruments, several recommendations are made that can improve biodiversity conservation policies. Stakeholder participation seems to be relevant in legitimating the instruments and can be a tool to resolve challenges of the instruments by identifying their problems and proposing solutions. Our results show that the combination of policy instruments improves the performance of individual instruments by generating synergies among them. Therefore, the perspective of a policy mix (Ring and Schröter-Schlaack, 2011) – that is, the interaction of different instruments when designing policies – can imply significant improvements in terms of effectiveness of instruments aimed at conserving biodiversity as well as the provision of ecosystem services. Moreover, it permits the identification of overlaps and conflicts between policies. Our findings show that few policies incorporate monitoring and control measures. Establishing these measures is considered necessary to reinforce the effectiveness of instruments and further assessing their implications in terms of efficiency, equity and legitimacy.

Appendix A.6.1 Structure of the questionnaire

The questions addressed to the research participants covered the following seven topics:

1. Whether they agreed (or not) with the category of stakeholder to which we have assigned them.
2. The reasons for which they participated in the PF of the natural park, both their individual reasons and the usefulness to participate regarding their category. In case they were not participating, they were asked whether they thought it could be useful to participate.
3. Why they thought the park managers promoted the establishment of the ECST and the PF.
4. Whether they thought the PF influenced positively the management of the protected area.
5. Whether they thought the ECST and the PF were effective, the reasons of their effectiveness and the issues that needed to be addressed to increase its effectiveness.
6. To rank at a scale from zero to five the character of the PF, with zero denoting "it is not" and five "it is a lot":
   - Whether the body was informative, meaning that the actions done or planned to be done were explained to participants in a unidirectional way.
   - Whether it was consultative, meaning that participants' opinions were requested without a joint discussion.
   - Whether it was interactive, meaning engaging participants in joint discussions leading to new opinions.
   - Whether it was decisive, meaning that agreed recommendations were implemented.
7. To rank at a scale from zero to five, with zero denoting "no influence/relevance" and five "a lot of influence/relevance" the different categories of stakeholders considering:
   - Their influence in the PF.
   - Their relevance in terms of attributed importance by others participants in decision-making.

Note that the four first questions of the above list have been presented to the research participants prior to the fieldwork through email. This was done to ensure that their answers would not be affected by the presence of other stakeholders.

Chapter 7. Conclusions

This thesis has analysed the implications of adopting ecosystem service and participatory approaches in the governance of protected areas. It includes five chapters addressing specific challenges associated with these two approaches.

Conclusions at chapter level

Chapter 2 presented a framework of relationship between biodiversity, ecosystem functioning, ecosystem services and has provided and illustrated a preliminary classification into five types of unwanted and avoidable indirect effects of biodiversity policy, namely rebound effects (biodiversity-spatial, biodiversity-incongruence, ecological, service and environmental). The analysis shows that some types of rebound relate to conflicts and the need for trade-offs between different types of biodiversity, or between certain types of biodiversity and certain ecosystem services. Identifying these potential rebound mechanisms will help in the design of effective policies for biodiversity conservation. Nevertheless, assessing the various types of rebound effects is a complex issue which requires the close collaboration between natural and social scientists to obtain a complete understanding of the various underlying natural and socio-economic mechanisms leading to biodiversity loss.

Chapter 3 illustrated the potential of using sociocultural valuation of ecosystem services for decision-making in protected areas. The study reveals the importance people attribute to the natural park of Sant Llorenç del Munt (Catalonia, Spain) for their well-being. The results show that the term ecosystem service is unknown by the general public and frequently misunderstood because it is related to human actions to improve nature. They further reveal that stakeholders are aware of the wide range of benefits provided by the natural park, of which habitat and cultural services are the most valued, especially those related to leisure activities. Stakeholders acknowledge, though, the importance of traditional rural activities linked to provisioning services in maintaining the Mediterranean cultural landscape and also as a means to diminish the risk of forest fires, to increase biodiversity and maintain traditional knowledge. The main characteristics of respondents that affect their sociocultural valuation of ecosystem services are age, education and place of residence. Priorities for management and potential conflicting viewpoints among stakeholders were assessed using sociocultural valuation which can improve protected area governance. Finally, this chapter concludes that using a mixed-method approach to value ecosystem services leads to a deeper understanding of socio-cultural values and permits to collect the multi-dimensional nature of value. Quantitative methods such as surveys provide a solid basis for comparison among ecosystem services to better inform decision-making processes whereas qualitative methods such as semi-structured interviews and non-participant observation offer rich information about how and why people value certain ecosystem services. They were used to inform the design of the quantitative valuation method identifying relevant ecosystem services and provide information about the social context in which the valuation took place.

Chapter 4 showed the usefulness of social network analysis as a tool to assess the representativeness of stakeholders in participatory processes. It compared the results of the social structure of the communication network of stakeholders of the natural park – analysed through social network analysis – with stakeholders' attendance to participatory processes – assessed through review of documents, interviews and non-participant observation. Results from our study indicate that the communication network of Sant Llorenç is fragile because of the few ties between stakeholders, which reflect a lack of trust and little knowledge exchange. The attendance to the participatory body of the park management shows that central stakeholders of the communication network are represented in it. However, lack of trust was found between some stakeholders that can hinder their involvement in the management of the natural park. Therefore, measures need to be adopted to improve the participatory bodies of the natural park to generate or rebuild trust among stakeholders, share information, provide support and increase cooperation between them. This chapter has demonstrated that social network analysis is a suitable tool to identify relevant stakeholders of different categories to support participatory processes.

Chapter 5 explored the political implications of participatory arrangements in the governance of Sant Llorenç. The reasons for stakeholder participation were assessed together with the issues addressed in the participatory body and the procedures and inclusiveness characterizing the body. In particular, regarding the reasons for promoting stakeholder participation, it has been found that participation is mainly enhanced by park managers in order to legitimate their decisions. Our results show that very often the prime driver for establishing participatory arrangements at the local level is the willingness to superficially cover or depoliticise previous conflicts and not normative choices embedded in governance rhetoric. Actually, a participatory body was created due to demands from the stakeholders involved in conflicts over the protection of the area and the urban development within and around its boundaries. These stakeholders were mainly conservationist and hiking groups from villages surrounding the area who wanted to preserve it from urbanization. The study found that the issues addressed in the Advisory Committee – the participatory body – are mainly technicalities of already accepted policies. The history of the creation of the natural park and its current governance dynamics have shown how the establishment of participatory arrangements instead of manifesting a shift towards more 'democracy' has actually led to the self-exclusion of social actors with key roles in the management of the protected area. In particular, the difficulty of reaching consensus among stakeholders attending the participatory body and the non-binding character of agreements resulted in withdrawal of certain stakeholders from the participatory process. The dynamics of the participatory body have further facilitated so-called "neoliberal biodiversity governance" by favouring the inclusion of actors with explicit economic motivations. The results have revealed that this goes together with notions that natural parks need to become self-sufficient entities which should be economically viable, generate their own economic resources, and have limited support from state resources. These findings confirm that participation needs to be understood as a political terrain, and suggest the need for evaluating, and if necessary reshaping, the conditions of participation.

Chapter 6 analysed the European Charter for Sustainable Tourism (ECST), a policy instrument aimed at promoting sustainable tourism by establishing a voluntary commitment between the managers of the protected area and all relevant stakeholders. Nine criteria were evaluated and potentially unwanted and avoidable indirect effects – known as rebound – were identified as potentially reducing the ECST's effectiveness. Results show that it is an effective instrument that engaged with different types of stakeholders in designing and implementing strategies to protect ecosystem services and biodiversity. However, the three dimensions of the equity criterion were evaluated negatively. The instrument is supported by most stakeholders except for the ones that stopped participating in it. Our analysis reveals that there is good complementarity among the different strategies and actions of the ECST and with other instruments implemented in the area. There is further little overlap or conflict among actions of the ECST and between them and other instruments implemented. A comparison with other policy instruments implemented in European countries was done together with the identification of different rebound effects of each instrument. Several recommendations were made that can improve biodiversity conservation policies, particularly the role of stakeholder participation in legitimating the instruments and in being a potential tool to resolve their challenges by identifying problems and proposing solutions. This chapter concludes that a perspective of a policy mix (Ring and Schröter-Schlaack, 2011) is necessary to improve the effectiveness of instruments aimed at conserving biodiversity as well as the provision of ecosystem services. It permits synergies among instruments and to identify overlaps or conflicts among them. It is further necessary to implement monitoring and control measures that not only assess the effectiveness of the instrument but also its performance in terms of efficiency, equity and legitimacy.

Synergies among insights from individual chapters

The reduction of rebound effects of biodiversity policies proposed in Chapter 2 can be improved by undertaking a sociocultural valuation of ecosystem services, as done in Chapter 3. This thesis has shown that biodiversity conservation policies can generate an unwanted and avoidable indirect effect on ecosystem services, namely service rebound. It can happen that this type of rebound can never be completely removed which leads to trade-offs. In cases of service rebound, knowing the importance attributed to ecosystem services by stakeholders through undertaking a socio-cultural valuation can help decision-makers to establish priorities in designing policy measures to reduce these indirect effects. However, adopting an ecosystem services approach in conservation policy has the risk of giving too much emphasis to ecosystem services at the expense of conserving biodiversity due to potential incongruences between these. For instance, promoting forest plantations focusing on CO_2 capture may reduce open areas which contribute positively to biodiversity. So while raising the importance of the benefits provided by ecosystems to human well-being may sometimes strengthen the motivation for biodiversity conservation policy, this is not always the case. Thus, biodiversity conservation has to be prioritized in protected areas while measures have to be implemented in order to minimize service rebound.

Both the valuation of ecosystem services undertaken in Chapter 3 and the assessment of rebound effects in Chapters 2 and 6 can lead to better insights if they are undertaken through participatory bodies attended by relevant stakeholders. Sociocultural valuation of ecosystem services can be a useful tool to complement decision-making processes that engage with stakeholders when implementing a policy instrument. This sociocultural valuation approach shows stakeholders priorities regarding ecosystem services which can facilitate the discussion about the effects of the policy instrument on them and further predict the reactions of stakeholders regarding the proposed policy. Moreover, the involvement of different types of stakeholders can help to identify rebound effects of biodiversity policy and propose measures to minimize them because different types of knowledge are represented in decision-making. For instance, stakeholders having certain knowledge about managing land, such as farmers or the forestry sector, can diagnose potential rebound effects of biodiversity policies.

The results of my thesis can illustrate this. Chapter 3 shows that tourism was not highly valued by stakeholders and visitors, which is not consistent with the strong emphasis on tourism by the policies implemented –as has been shown in Chapters 5 and 6. Actually, many stakeholders identified tourism as an activity that should be better managed; particularly, the number of visitors coming to the natural park should be controlled and even reduced. Moreover, stakeholders claimed for enhancing agro-silvo-pastoral sustainable practices as a way to maintain cultural landscapes and avoid challenges resulted from the abandonment of these traditional activities. However, results found that the European Charter for Sustainable Tourism, despite aiming to address many challenges of the natural park, prioritized the promotion of tourism. Furthermore, many of the actions targeting to enhance agro-silvo-pastoral sustainable practices were not undertaken. In addition, chapter 5 has pointed out that the dynamics of the participatory bodies of the protected area have resulted in an over-representation of stakeholders related to tourism with the consequent self-exclusion of other relevant sectors. This situation can lead to a negative feedback loop by further strengthening tourism activities and increasing the already high number of visitors. Moreover, it might result in designing policies focused too much on the demands of this sector and forget about the realities of other stakeholders of the protected area, as is already happening. It might further result in trade-offs among ecosystem services and with biodiversity conservation.

This thesis has found that the natural park of Sant Llorenç has the conditions to promote institutional arrangements for including stakeholders in decision-making. It is particularly relevant, as studied in chapter 4, that Sant Llorenç has a social network which facilitates stakeholder involvement in governance. We argue that the European Charter for Sustainable Tourism can be this participatory body to generate dynamics of involvement of stakeholders in discussing policy problems and engage with decision-making. Nevertheless, it lacks the indispensable representativeness of stakeholders and the willingness of park managers to implement all the actions approved to deal with challenges of the natural park, as found in Chapter 6.

This thesis has shown that an ecosystem service approach and stakeholder participation in governance of protected areas can be a central component in connecting biodiversity conservation with people's well-being and promote environmental justice in protected areas governance. However, they need to be evaluated; otherwise they may promote an overemphasis on ecosystem services at the expense of biodiversity conservation in cases of incongruence between them and develop participatory arrangements that exclude relevant stakeholders from decision-making and promote neoliberal biodiversity governance.

Further research

There are many issues that deserve more attention in future research. As a methodological issue, it is necessary to clarify the foundations of the sociocultural valuation of ecosystem services. The use of the concept ecosystem service has been initially linked to the necessity to express the value of ecosystems in monetary terms in order to reflect them in prices and national accounts. The aim of sociocultural valuation is different, being focused on raising knowledge and awareness of the contribution of ecosystems to human well-being, together with highlighting the different sociocultural preferences among stakeholders. It can be a tool to facilitate the incorporation of stakeholders' opinions regarding ecosystem services in decision-making and thus assist in their participation in the governance of protected areas. Research is thus needed that strengthens the potentialities of the ecosystem services approach to integrate stakeholder viewpoints in decision-making and promote environmental justice. Another issue that deserves further attention is the shift towards "neoliberal conservation" approach in the last decade. In particular, studies need to be undertaken to assess its precise consequences in terms of biodiversity protection, also in comparison with other instruments and institutions. This thesis has addressed the implications of this shift in terms of environmental justice by showing an associated problem of exclusion of certain stakeholders in participatory processes supporting governance. This leads then to the question of what type of formal participatory arrangement is best and which informal network structures can solve this problem. Finally, studies might address which type of policy instruments or combinations of instruments are most effective in limiting rebound and so assure effective biodiversity conservation.

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