Etude des compromis et synergies entre services écosystémiques et biodiversité : Une approche multidimensionnelle de leurs interactions dans le socioécosystème des Alpes Française

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Etude des compromis et synergies entre services écosystémiques et biodiversité

Une approche multidimensionnelle de leurs interactions dans le socio-écosystème des Alpes françaises

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Il y aurait des choses

On y trouverait, pêle-mêle, un ordinateur, des balles, une bouilloire, un baudrier d'escalade, les logos de R, Mendeley *et* ArcGIS *(d'accord, et* Powerpoint *aussi), plusieurs paires de baskets, un enregistreur, des toupies, une nappe rouge à pois blancs, une slack line, des tonnes de pochettes transparentes remplies de biblio, un vélo, quelques rush de tournage, des gâteaux cachés dans les tiroirs, des playlists aux influences variées, le Kangoo, des dossiers pleins de schémas, de brouillons, de corrections, de retranscriptions, d'analyses plus ou moins abouties, de documents en versions 'draft/finale1/2/3/_SL/last/'…*

Il y aurait des lieux

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TABLE OF CONTENTS

INTRODUCTION

CHAPTER I - BIOPHYSICAL ASSESSMENT

CHAPTER II - INFLUENCE NETWORKS

CHAPTER III - POLICY MIX ANALYSIS

CHAPTER IV - EPISTEMIC COMMITMENTS

DISCUSSION

APPENDICES

INTRODUCTION

Introduction

I. Conceptual background: On the needs and methods for considering social and ecological interactions in cultural landscapes

"*Cultural landscapes are at the interface between nature and culture, tangible and intangible heritage, biological and cultural diversity – they represent a closely woven net of relationships, the essence of culture and people's identity*."
(Rössler 2006:334, in Plieninger et al. 2014)

Cultural landscapes have been shaped through long-lasting and dynamic interactions between human, organised in societies, and nature, constituting their surrounding and supporting biophysical frame (Schaich et al. 2010). Changes both at global and local scales (e.g. climate change or polarization of land uses, respectively) are inducing major transformations in cultural landscapes worldwide (Plieninger et al. 2014), driven by a fundamental decoupling of sociocultural and ecological components (Fisher et al. 2012). These changes are iconic of the new "Anthropocene" geological era we have entered (Steffen et al. 2007), the first era dominated by such a human footprint on the biosphere that biophysical processes currently undergo severe threats putting at stake irreversible environmental and social changes (Rockstrom et al. 2009).

In this context where the future of many cultural landscapes appears uncertain (Plieninger et al. 2014), addressing the determinants, modalities and impacts of ecosystem management is both a challenge and a necessity to sustain human well-being (MEA 2005a, Stevenson 2011). In this endeavour, conceptual advances are required regarding the objects of study and the methods employed to assess them, together with empirical progresses that would provide practical knowledge for environmental resource management at various scales. My PhD project aims at exploring the French Alps landscapes in this perspective, with the underlying motivation that the different domains of knowledge I interweaved could contribute to a more comprehensive and transdisciplinary understanding of the area.

A. Social-ecological systems – Formalizing the links between people and nature

The assessment of landscape dynamics, and in particular European *cultural landscapes* (EEA 2010), requires the joint consideration of the social and ecological processes that have shaped them through time. The concept of *social-ecological system* has been proposed to represent these intimate interconnections between humans and ecosystems, which additionally appear at nested and interacting scales (Ostrom 2009). They have also been called also called '*Coupled Human and Natural Systems*' (Liu et al. 2007). At the conceptual level, a given social-ecological system can be defined as a "system that includes societal (human) and ecological (biophysical) subsystems in mutual interactions (Gallopin 1991) and thus captures interactions between ecosystems, biodiversity and people" (Harrington et al. 2010). Interactions occur both within each of the ecological and social sub-systems and also between them, inducing complex feedbacks (Anderies et al. 2004, Folke 2006).

Figure 1 proposes a schematic vision of a conceptual social-ecological system, adapted from Martín-López et al. (2009). In the social system, people dynamically interact and are organised through scales according to the institutions (i.e. the set of shared rules, including the economy) that frame their behaviour (Harrington et al. 2010). In the ecological system, organisms (plants, animals, micro-organisms) are organised according to their functional characteristics, to the abiotic setting and to their dynamics in space and time (MEA 2005a), from local scale to landscapes and biomes. Social systems interact with ecosystems at different scales through management and resulting modifications of ecosystems. In turn, ecosystems supply resources and functions that lead to social benefits (the *ecosystem services*, see next section) or constraints (sometimes called *ecosystem dis-services*, Lamarque et al. 2011a).

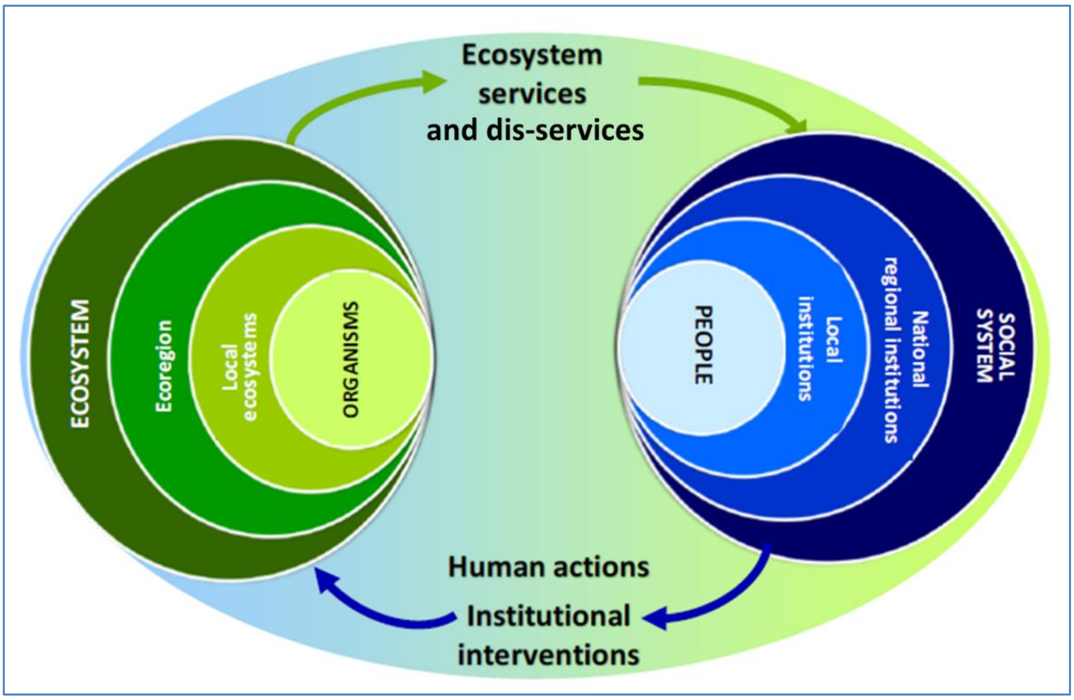

Figure 1: Conceptual diagram of a social-ecological system. Adapted from Martín-López et al. (2009)

B. Ecosystem services (ES) - At the interface between social and ecological systems

1. The need for a new concept

At the interface between the social and the ecological systems, *ecosystem services* (hereafter ES) have been proposed to make explicit "the benefits people obtain from ecosystems" (MEA 2005a). They are defined as "the direct and indirect contributions of nature to human wellbeing" (TEEB 2010) and stress human dependency on natural processes (Diaz et al. 2006, Diaz et al. 2015). The rationale supporting the ES concept is to propose an alternative to classical conservation arguments that failed at stopping, or even limiting, the human-induced damages on ecosystems and biodiversity losses worldwide (Mace et al. 2010).

The originality of the ES concept is to highlight that sustainable management of ecosystems is not a luxury (Granjou & Mauz 2011), but rather a vital necessity to sustain basic human needs and further to contribute to individual and social well-being (Mainka 2005).

Early mentions of the concept date back to the 1970s, under the terminology 'nature's service' (Westman 1977). Rapidly, the term of *ecosystem service* was seized by the scientific community (e.g. Ehrlich & Mooney 1983) as a mean to raise awareness of the global biodiversity loss and ecosystem degradation (Lamarque et al. 2011b). A growing body of literature has since then made use of the concept (see the quantitative reviews by Vihervaara et al, 2010, Lautenbach et al. 2013, Abson et al. 2014). Its influence has spread far from the academic sphere into the policy and economic fields with as major milestones two world-wide initiatives to assess and value the contributions of ecosystems to human wellbeing: the *Millennium Ecosystem Assessment* (MEA) in 2005 and *The Economics of Ecosystems and Biodiversity* (TEEB) in 2010. Thus, in some 30 years, ES turned from a metaphoric to a heuristic concept (Abson et al. 2014) and further to a "concrete, tangible and measurable" object (Barnaud & Antona 2014). Iconic of this reification into an explicit decision and policy tool (de Groot et al. 2010) is the initiation of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES http://ipbes.net/work-programme.html) in the early 2010s, which is structured around four major objectives: biodiversity and ES assessments, knowledge generation, capacity-building, and policy support (Diaz et al. 2015).

2. ES – Some definitions

The ES have been described as a link between "ecological structures and processes created or generated by living organism and the benefits that people eventually derive", all these elements being organised as a descending *cascade* (Haines-Young & Potschin 2010). To account for the feedbacks from the social system on the ecological one, authors have proposed to close the loop through an ascending *stairway*. It represents the influence of policy, land planning and management choices, which rely on people's preferences and on practical intervention measures (Spangenberg et al. 2014).

Due to their interface position in the social-ecological system, ES are fully described according to three constitutive facets accounting for each sub-system and for the interconnection of both.

i) *Potential supply*: the ecosystem potential "capacity to supply services" (Bastian et al. 2012), considering its geophysical and ecological characteristics in the current land cover matrix,
ii) *Demand*: "the amount of service desired by society" (Villamagna et al. 2013), irrespective to the ability of the ecosystem to fulfill this desire,
iii) *Actual supply*: the actual encounter of demand and potential supply, also accounting for external drivers as legislation or economic constraints.

ES are usually classified in three categories:

i) *Provisioning ES*: the goods obtained from ecosystems, such as food, freshwater or timber,
ii) *Cultural ES*: the intangible benefits people obtain from ecosystems through outdoor recreation, landscape aesthetic experiences or presence of iconic species,
iii) *Regulating ES*: the benefits obtained from the ecosystem functioning such as maintain of soil fertility, biotic contribution to erosion control or pollination.

A fourth category of *supporting ES* has been proposed in some classifications (MEA 2005a) to account explicitly for the biophysical cycles essential for the other services to be supplied. Despite the acknowledged necessity of maintaining these processes, issues of *double-counting* regarding what would be *indirect* services (relative to those leading to a *direct* human benefit) led to their exclusion as such from ES assessments, as "they are not ends in themselves" (Wallace 2007). The processes encompassed in the initial supporting category have been identified as ecological functions, or alternatively as an *ecological integrity* indicator that can be assessed jointly with the other three ES categories (Lamarque et al. 2011, Burkhard et al. 2012).

It must be noted that *biodiversity* (i.e. "the variability among living organisms from all sources […] and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (MEA 2005b)) as such is not an ES, as it does not induce a direct gain in human wellbeing. Additionally, the links between biodiversity and ES are complex, non-linear and dynamic (Haines-Young and Potschin 2010) and remain incompletely captured to date (Kremen 2005, Cardinale et al. 2012). Rather, biodiversity is to be considered as a necessary support for all ES and further as a prominent determinant of ecosystem adaptive capacity and resilience to global changes (Cardinale et al. 2012). Thus, biodiversity as a conservation objective is not to be replaced by ES, and the two concepts should rather complement and support each other in the objective to maintain dynamic and functional ecosystems (Chan et al. 2006, Schröter et al. 2014). Additionally, further understanding remains to be gathered on the determinants, generality and strength of spatial congruence between multiple ES and biodiversity.

Ecosystems can provide multiple ES, although their supply and demand will vary both in time and space (Fisher et al. 2009). A *synergy* represents a positive repeated co-variation between two ES, while a *trade-off* stands for a negative association (Mouchet et al. 2014). Many studies assessed i) binary relationships among various ES and ii) areas combining high (respectively low) levels of multiple ES, i.e. *hotspots* (respectively *coldspots*) (e.g. Egoh et al. 2008, Anderson et al. 2009). However, accounting for the joint variation of multiple ES is a complex task still under-addressed (Chan et al. 2006, Tallis et al. 2008, Bennett et al. 2009, Reyers et al. 2013). Assessing *bundles* of ES, i.e. consistent associations of ES over time and/or space (Raudsepp-Hearne et al. 2010), has been proposed as a relevant solution to increase understanding of common ecological and social determinants. This is indeed required to improve the predictability of management option impacts (Mouchet et al. 2014).

To date, despite progress on both the conceptualisation of ES and the understanding of interlinkages among ES and between ES and biodiversity, few studies have linked i) insights from conceptual frameworks describing ES consistent associations with ii) an explicit accounting of their three facets (Crouzat et al. *submitted*). Uniting both appears a promising direction to obtain a more comprehensive understanding of constraints and opportunities linked to ES bundle management.

Figure 2 shows the interface position of ES as my work will refer to, accounting for the various directed influences shared with the ecological and social systems.

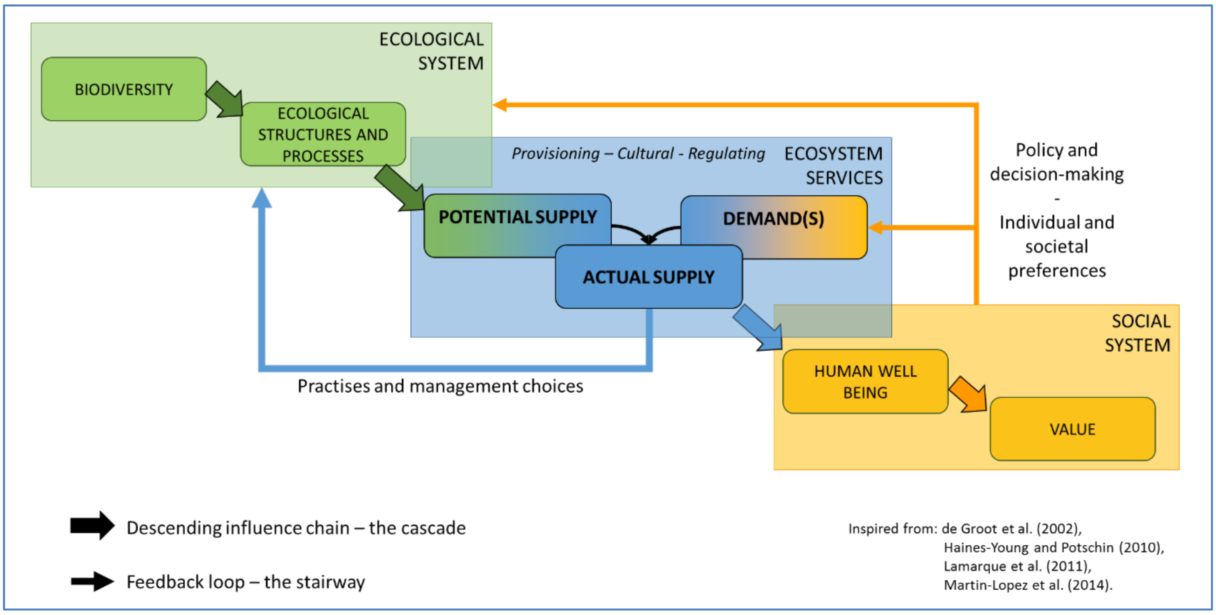

Figure 2: The ES facets (potential supply, demand, actual supply) at the interface of the ecological and the social system. Descending influences from the ecological system to the social system are usually referred to as 'the ES cascade' and are complemented by influences in the ascending direction creating 'a stairway'.

3. Considering multiple value-domains for assessing ecosystem services

ES science deals with a 'hot' concept that is neither stabilised nor consensual (Barnaud & Antona 2014). Efforts are made toward common definitions (e.g. Fisher et al. 2009, Lamarque et al. 2011b) and toward a more accurate capture of ecological and social processes in ES assessments. But despite these progresses, a fundamentally irreducible scientific *uncertainty* remains, due to the inherent complexity of the systems targeted (Pielke 2007). Additionally, *controversies* regarding the ES concept itself remain topical and include, among others, ethical considerations on the human-nature relationship, issues linked to valuation methods and risks of nature commodification (recently addressed by Schröter et al. 2014). In the context of an increased uptake of the concept for policy and management purposes (Jax et al. 2013), there is thus a risk that what is actually a *science in-the-making* would be taken as *ready-made science* delivering a "unique and complete understanding of a phenomenon" (Barnaud & Antona 2014). While in Chapter IV and the General Discussion I will explore the major pros and cons of the ES concept, some important statements should be mentioned here regarding the normative dimension of ES assessments.

ES exist only if someone, i.e. a human being, demands and benefits from them. The concept is thus embedded in an anthropocentric vision of the world (Luck et al. 2012, Fisher & Brown 2014), i.e. a separation of 'nature and culture' following Descola's words. This induces that using ES to explore our relation to nature is not only manipulating a *descriptive* framework but also choosing a *normative* concept (Abson et al. 2014). Environmental assessments are performed to quantify and/or qualify "the value" of the ES used, protected or impacted by the various stakeholder groups of a given social-ecological system.

Three value-domains have been proposed for ES assessments (Martín-López et al. 2014):

i) *The Biophysical value-domain*: this domain accounts for the state of an ecosystem and for its ability to supply ES, measured with ecological indicators and biophysical units (de Groot et al. 2010). Numerous modelling methods have been developed to quantify ES values based on biophysical information, for instance, with increasing complexity, statistical models (e.g. Brus et al. 2011 – tree species distribution), empirical models (e.g. Bosco et al. 2009 – erosion control), macro-ecological models (e.g. Civantos et al. 2012 – species distribution), phenomenological models (e.g. Schulp et al. 2014 - pollination) and trait-based models (e.g. Lavorel et al. 2011 – grassland agronomic value) (Lavorel et al. 2014). Models are often based on proxy data (i.e. indirect estimates), resulting in the need for a careful attention to the actual meaning and level of confidence associated with mapped outputs (Eigenbrod et al. 210).

ii) *The Socio-cultural value-domain*: this domain stresses the moral, ethical and cultural motivations to value nature (Martín-López et al. 2009). Stakeholders have been proved to hold varying values toward environmental resources (e.g. Hicks et al. 2013, Iniesta-Arandia et al. 2014), leading to differing perceptions of the social-ecological system (e.g. Lamarque et al. 2011a, Gos & Lavorel 2012). Methods to elicit these motivations examine "the cognitive, emotional and ethical arguments, preferences and demands expressed by people towards nature" (de Groot et al. 2010). Among others, they include participative methods such as focus groups, mental mapping, ranking or citizen juries (Chevalier & Buckles 2008, Chan et al. 2012b). These methods lead to an explicit representation of the system as it is perceived by different stakeholder groups, which can be seized to collectively discuss the current and future management of a given territory. Such collective processes potentially create social learning and can be the base for a co-adaptive management of environmental resources (Armitage et al. 2009).

iii) *The Economic value-domain*: this domain conceives the value of ES in terms of *utility*, i.e. relatively to the satisfaction experienced through the consumption of a good (TEEB 2010). Different methods have been developed to obtain ES 'Total Economic Value' (e.g. market prices, value transfer, contingent valuation, willingness to pay/to accept), which encompasses direct use, indirect use, option and existence values (Pearce and Turner, 1990, in Martín-López et al. 2009). Yet, choosing the method most appropriated to fit i) the ES assessed, ii) the scales of focus and iii) the questions addressed still remains challenging and calls for further methodological progresses (Bateman 2011, Atkinson 2012, Brouwer et al. 2013, Kumar 2013).

Overall, the ES concept has been proposed to engage diverse stakeholders against biodiversity loss and ecosystem degradation, including policy-makers. In this context, mapping methods have been highlighted as particularly appropriate to support understanding and communication of assessment outputs to a diversity of stakeholders (Martínez-Harms & Balvanera 2012).

The current neo-classical economic system in which the ES concept arose tended to favour the economic value-domain in ES assessments (Gómez-Baggethun & Ruiz-Perez 2011). Alternative biophysical and socio-cultural value-domains can also be relevantly mobilised in its stead or as equal complements, even though calls for their increased consideration remain to be further answered in practical assessments (Chan et al. 2012a, Martín-López et al. 2014).

C. Endorsing the non-neutrality of ES science

ES science has made good progress in the last decades towards interdisciplinarity by proposing concepts, methodologies and assessments that can be jointly grasped by natural and social sciences, even though progresses are still possible to fully develop a social-ecological system approach (Reyers et al. 2013). Meanwhile, practical studies focused on environmental assessments and decision-making seem to dedicate a generally low attention to more *purposive* aspects, i.e. to the 'level of meaning' that encompasses ethics, values and philosophy (Hadorn et al. 2006, Reyers et al. 2010).

Overall, ES science is not a neutral monolith disconnected from values, judgments and choices. There is thus a need for ES scientists to "find their place" at the interface between science and society (Donner 2014). Multiple postures can be adopted depending on whether researchers mainly pursue understanding, governance or advocacy (Pielke 2007, Coreau et al. 2013, Donner 2014). Options range from a pure scientific posture absolutely disconnected from social concerns to intermediate engagement facilitating the inclusion of advanced knowledge in decision-making, and further to public advocacy explicitly defending a particular stance. Such options describe what is called the *epistemic commitment* of a researcher and more generally of any stakeholder wanting to use knowledge to support or to guide a choice (Arpin & Granjou *in press*). Each commitment is linked to a specific science-society contract that may be i) conscious or not and ii) made explicit or not.

To progress toward more transparent and explicit relationships between all stakeholders, there is a growing call to formalise and communicate the values and ethics underlying projects using the ES concept, i.e. there is a need for an explicit assessment of epistemic commitments of all stakeholders involved in such projects (Pielke 2007, Donner 2014). ES scientists should therefore further engage with the *axiological* dimension of their work, i.e. with the value background they interweave with their scientific advances (Weinberg 1970). Indeed, "once we admit that environmental problems may reflect our own culture and attitudes as much as a scientific or technical problem, we have greater scope for possible responses" (Ludwig et al. 2006, in Reyers et al. 2010).

D. Governance of ecosystem services – Exploring formal institutions around ES

For ES to articulate on the one hand natural resources and sensitivity of ecosystems with on the other hand needs and impacts of humans (MEA 2005a, Steffen 2009), social arrangements are required to allocate resources and control uses. This is what *governance* is about, being more formally defined as "all the institutional arrangements and processes aiming at identifying and enacting collectively acceptable principles" (Primmer & Furman 2012). Governance concerns all actors, from governmental, inter-governmental, and nongovernmental organisations, from the private sector and from civil society (Greiber & Schiele 2011).

The various rules that govern the behaviour of stakeholders are called *institutions* (Pahl-Wostl 2009). They include i) *formal* institutions, linked to the official channels of regimes empowered, that are codified and enforced by legal procedures (Greiber & Schiele 2011) and ii) *informal* institutions that are "socially shared rules such as social or cultural norms" (Pahl-Wostl 2009). While informal institutions respond to slow dynamics expressing profound structural changes, formal policy instruments can be more rapidly and explicitly adapted to effectively manage environmental resources (Armitage et al. 2008). Sustainable management of ES could thus target as a first step policy instruments.

To address the complexity of environmental management, *policy mixes* are put forward as they enable integrating concerns from multiple sectoral policies. A policy mix is defined as “a 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” (Ring and Schröter-Schlaack 2011). A policy mix does not necessarily support the joint supply of all ES and biodiversity aspects and usually includes multiple instruments with specific targets, which can be complementary, synergistic or conflictual.

Numerous criteria have been proposed for the design and evaluation of policy mixes regarding their environmental impacts and benefits. They usually consider *environmental effectiveness* (i.e. the effects of the instrument on environmental quality) and *economic efficiency* (i.e. the cost/benefit balance linked to the application of the instrument) (Ring & Schröter-Schlaack 2011). These traditional criteria can be complemented by drawing attention to *fairness*, *justice*, *coherence* with the legal and institutional systems or *precaution* (regarding serious or irreversible consequences that need to be avoided). Additionally, articulation of instruments within the policy mix is addressed through the identification of the positive *complementarities* enhancing global effects and the negative *overlaps* and *conflicts* undermining environmental effectiveness. Recently, authors have added to these criteria the analysis of *rebound effects*, i.e. the positive and negative collateral effects of policy instruments on untargeted environmental aspects (inspired from Maestre et al. 2012).

There seems to be a general discrepancy between the announced objective of ES assessments to provide effective governance options and the apparent lack of practical consideration of institutions in these assessments: actual accounting of ES in governance is only emerging (Carpenter et al. 2009). There is thus “an urgent societal challenge” to design policies that can protect and enhance ES supply (Reed et al. 2013). To date, this remains conditional to conducing ES assessments that further consider “existing policies and the institutional context” as a key element in their approach of social-ecological system, “together with the ecological and socio-economic context of ecosystem service use and management” (Primmer & Furman 2012). To my knowledge, no explicit analysis of a policy mix following an integrative set of criteria has yet been proposed to assess ES governance.

II. Context – The European CONNECT project and its French Alps case-study

I developed my PhD project, entitled Addressing trade-offs and synergies among ecosystem services and biodiversity – A multi-dimensional approach of their interactions in the French Alps social-ecological system, in the context of the CONNECT project. The overarching objective of this European ERA-Net BiodivERsA project (2012-2015 http://www.connect-biodiversa.eu/) is to investigate the relationships between biodiversity and ES. Indeed, there remains uncertainty about the strength and generality of spatial congruence among biodiversity and ES, which makes difficult to propose general rules for sustainable natural resource management (Tallis et al. 2008, Maes et al. 2012, Zupan et al. *submitted*). The CONNECT project proposes a theoretical and empirical investigation of the relationships between ES and biodiversity over Europe, relying on the hypothesis that improved insights will help sustaining both ES supply and biodiversity conservation through an adequate design of management strategies and policy tools.

The CONNECT interdisciplinary consortium consists of five partners representing a broad range of disciplines relevant to ES science and to addressing this challenging question. Each partner is responsible for one of the objectives targeted by the interrelated work packages (WP), although it contributes as well to the other WP (Figure 3).

- WP1 aims at relating biodiversity facets (taxonomic, phylogenetic and functional diversity) and important ecosystem functions associated with ES supply.
  ➔ *Laboratoire d'Ecologie Alpine, Université Joseph Fourier (LECA - CNRS)*
- WP2's objective is to develop ES modelling methods of intermediate complexity at regional scale to analyse interactions among ES and biodiversity.
  ➔ *Institute for Environmental Studies, VU University Amsterdam (VU-IVM)*
- WP3 contributes to the development of improved nonmarket ES valuation techniques, paying particular attention to the spatial context and the underlying ecological structure and processes.
  ➔ *Helmholtz Centre for Environmental Research (UFZ)*
- WP4 coordinates five case studies at regional scale representing typical cultural European landscapes. Each case study develops a stakeholder dialog to inform the regional relationships between ES and biodiversity and to reveal the role of current policies. Additionally, a cross-cutting assessment over European Natura2000 and High Natural Value farmland areas is carried out to provide a European overview and context to the regional case studies.
  ➔ *Lund University (ULUND)*
- WP5 integrates findings from WP1-4 to propose guidelines for designing efficient policy instruments that sustain both ES supply and biodiversity conservation.
  ➔ *Universidad Autònoma de Barcelona (UAB)*
- WP6 is in charge of managing the project and coordinating the dissemination of its results.
  ➔ *Institute for Environmental Studies, VU University Amsterdam (VU-IVM).*

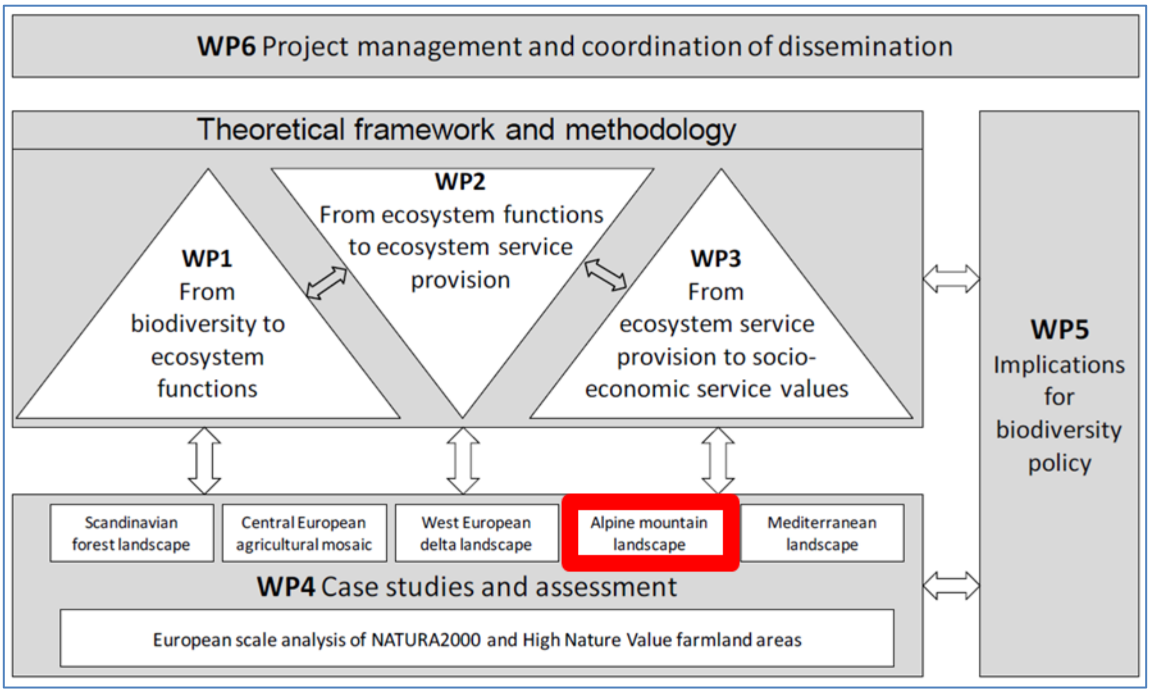

Figure 3: Overall organisation of CONNECT project in 6 work packages (WP), and highlight on the French Alps case study within WP4.

My PhD work contributed to WP4, which uses case studies to test methods and findings for operational environmental management. In particular, LECA was responsible for the alpine mountain landscape assessment, with a specific focus on the French Alps area (Figure 3).

III. The French Alps as a social-ecological system

The French Alps are a mountain region covering approximately 50 000 km² in the western part of the Alpine arc (Figure 4). They expand over two NUTS-2 levels ("régions" Rhône-Alpes and Provence-Alpes-Côte d'Azur) and nine NUTS-3 levels ("départements") that encompass 21.4% of the total area covered by the Alps over eight countries in the centre of Europe, for a population weighting 17.5% of the whole alpine population (2 453 600 inhabitants in the French Alps in 2007) (SPCA 2010).

Altogether, the diversity of biophysical and human uses is responsible for the high variety of biodiversity, ecosystems and ES across the entire area (Tappeiner et al. 2008, Crouzat et al. in review). In the following sections I describe the general features of the French Alps social-ecological system which need to be considered to get a first contextual approach of these areas of high cultural and ecological importance, in the context of joint global and local changes.

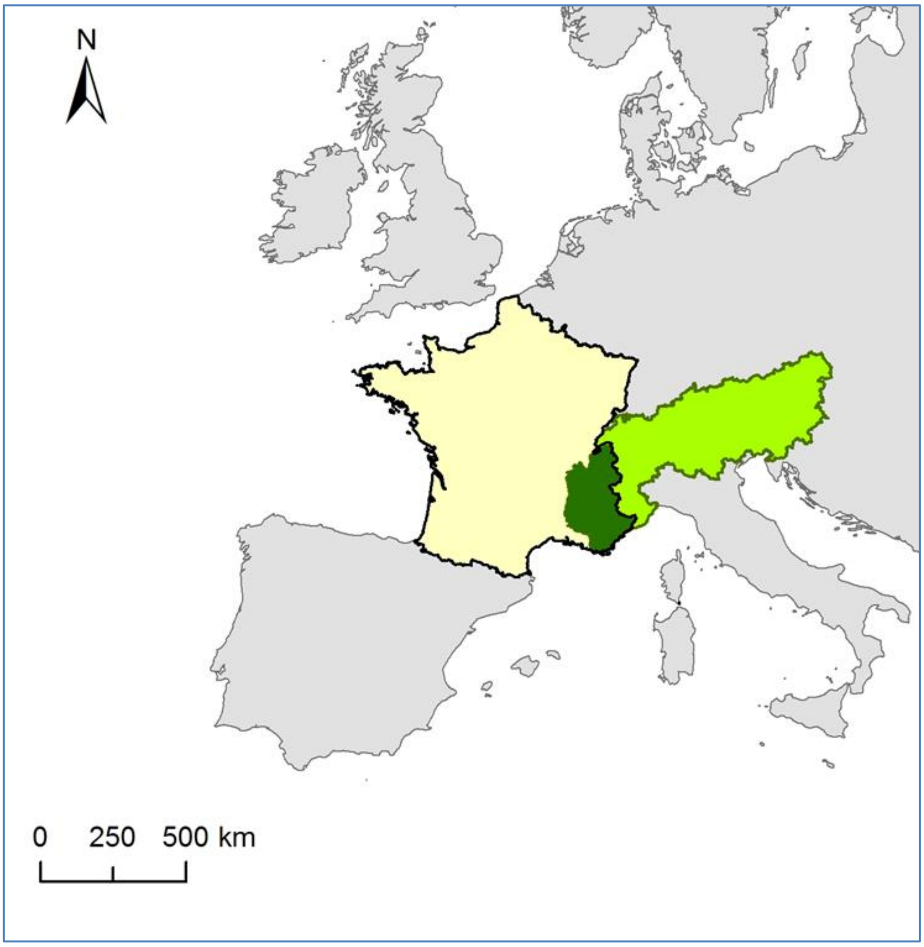

Figure 4: The French Alps (dark green) within France (yellow) and the Alpine massif (light green).

A. Main pressures on natural and semi-natural areas

Land use change is a prominent alpine driver of biodiversity loss, through an extensive urban and suburban development and an increasing tourism demand (Walzer et al. 2013). This increases i) fragmentation of the territory (human infrastructures), ii) demand for recreation and other amenities (scenery...) and iii) pressure on existing agricultural and forest management strategies. Modification of agricultural management (i.e. land abandonment in extensive areas and intensification in favourable areas) also modifies plant and animal biodiversity, as well as landscape quality and position of treelines (MacDonald et al. 2000, Tasser et al. 2007).

Climate change is as well recognized as a threat for biodiversity and landscape quality, as the Alps have undergone a temperature increase of around + 2 °C between the late 19th and early 21st century, more than twice the rate of average warming of the Northern hemisphere (Engler et al. 2011). In particular, the altitudinal and meridian gradient in the increase of temperatures threatens alpine species that face both a restriction in their favourable habitats and an increased competition from more generalist species. Climate change is also foreseen to modify water cycles in temporality and quantities, leading to increased pressures on ecosystems (e.g. from accentuated summer droughts) and related ES, in particular those linked to the agricultural and tourism sectors (EEA 2009).

Other threats like *biological invasions* or *pollution* (including N deposition) pose more limited risks, though present in some areas.

B. General characteristics at sub-regional scale

The following section summarizes important characteristics at sub-regional level accounting for biophysical features, current land uses and related social trends. Usual altitudinal and land cover variables (Figures 5.A and 5.B, respectively) were enriched by the description of the alpine social-ecological system as proposed by the DIAMONT project (2004-2008, Interreg IIIB-Project, Alpine Space Program). The objective of this project was to contribute to a complete and unified picture of the whole Alps based on common economic indicators, social and cultural trends as well as on ecological data. From the very interesting insights from this project, I propose two illustrations over the French part of the massif that i) characterise regions according to their local dynamics of development (Figure 5.C), and ii) highlight the overall human impact on the environment, also called *hemeroby* (Figure 5.D). The aggregative index of hemeroby accounts for the intensity and direct impacts of human activities on main land use types. It does not consider indirect impacts from global pressures as climate change nor pressures with a spatial dependency effect (e.g. upstream/downstream dynamics). Land use types unaffected by local human impacts are assigned a low value (1, e.g. glaciers, virgin rocky areas) while semi-natural and cultivated areas obtain intermediate values (2 - 5, e.g. forests, pastures, permanent crops) and completely artificialized areas are given a high value (7, e.g. densely built-up settlement areas). The final value is calculated by weighting the areas of different land use types at the municipality level.

Information from these four sources (altitude, land covers, dynamics of development and hemeroby) has been visually extracted along latitudinal and longitudinal gradients (Figure 6) and is further presented below, expanding on the description proposed by DIAMONT outputs (I refer interested readers to the inspiring atlas "Mapping the Alps" related to this project (Tappeiner et al. 2008)).

It should be mentioned that the perimeter of interest in this manuscript includes the territory of all nine "départements" concerned by the Alpine Convention perimeter (cf. next section on governance). Although the Convention is more restrictive in its understanding of “alpine territories”, several statistics, datasets and governance instruments related to the administrative delineation of “départements”. Thus, we decided to keep an extended perimeter in our analyses (52 149 km² vs. the ‘official’ 40 801km²) (Figure 5.A purple delineation).

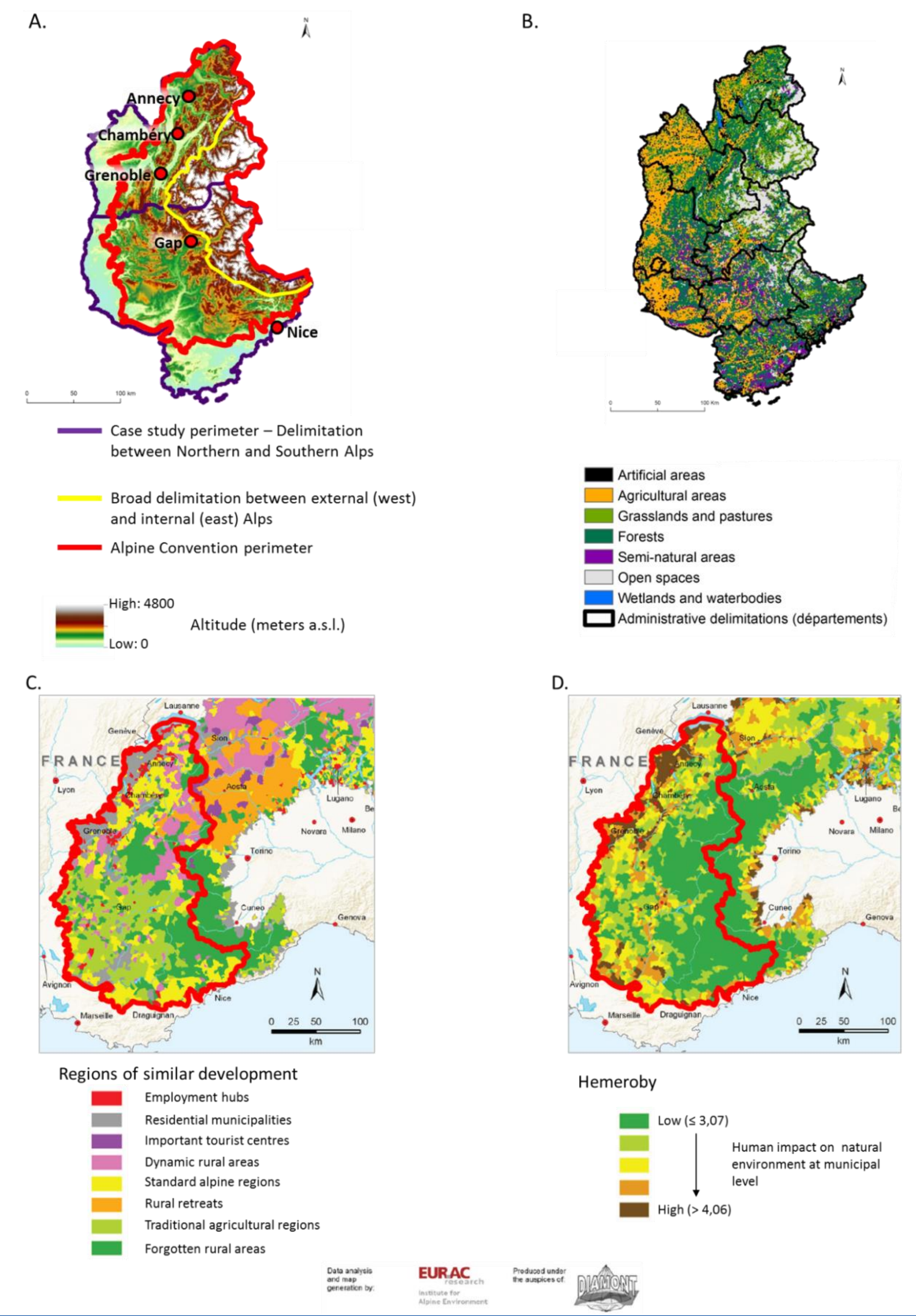

Figure 5: Some characteristics of the French Alps area:
A. Altitude (meters a.sl.) - Broad delimitation between Northern and Southern Alps (purple) and between internal and external Alps (yellow).
B. Main land cover categories according Corine Land Cover 2006. Black delineation symbolises the administrative boundaries of "départements".
C. Typology of the Alps (zoom on the French part), based on economic, environmental and social aspects (extracted from Tappeiner et al. 2008). The red outline represents the Alpine Convention perimeter.
D. Hemeroby in the Alps (zoom on the French part) (extracted from Tappeiner et al. 2008). The red outline represents the Alpine Convention perimeter.

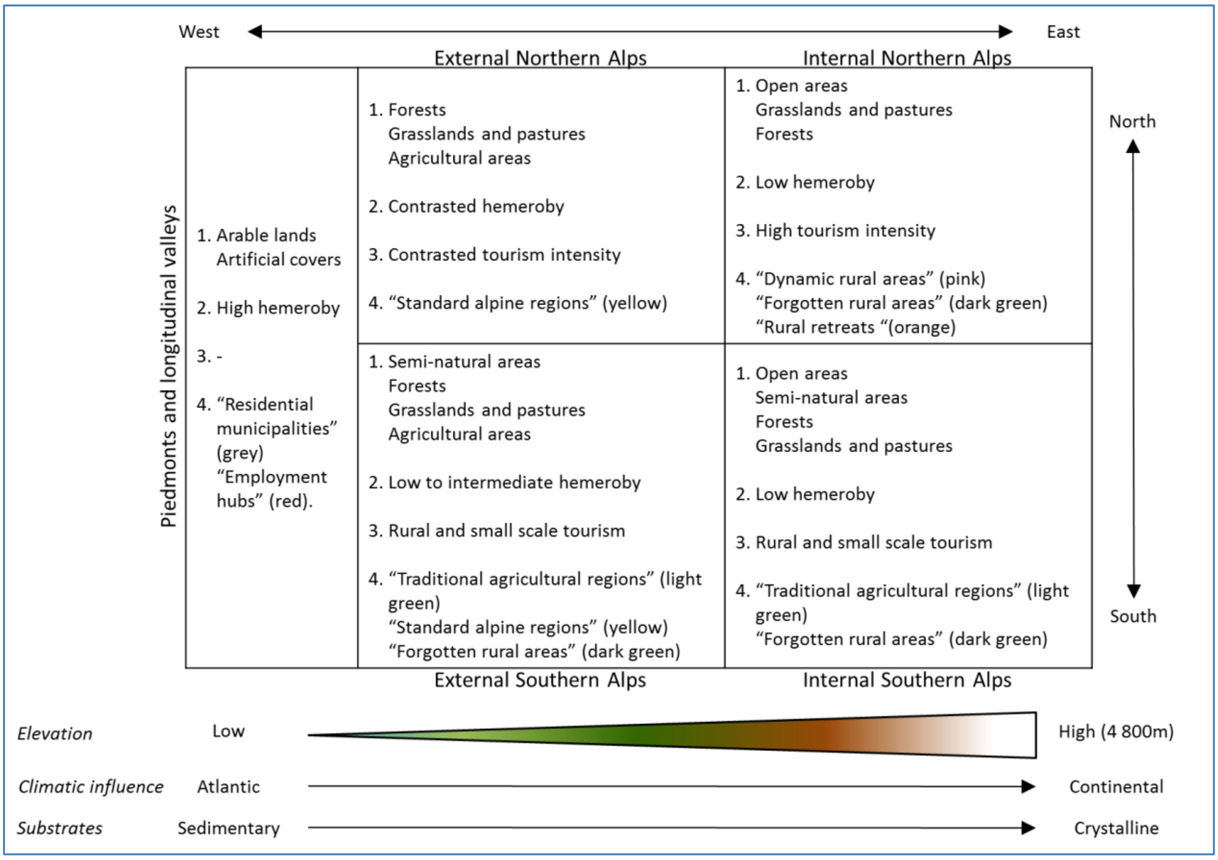

Figure 6: Synthetic overview of important characteristics of French Alps sub-regions along latitudinal and longitudinal gradients:
1. **Main land cover categories extracted from Figure 5.B;**
2. **Intensity of human impacts on natural areas (hemeroby) extracted from Figure 5.C;**
3. **Dominant form of tourism;**
4. **Main types of development extracted from Figure 5.D.**

The French Alps differ from the whole massif main orientation by a meridian axis (north/south). This orientation implies a large latitudinal climatic and vegetation gradient, with historical consequences on social dynamics and economic activity. It explains the usual division of the whole area in two main regions: the Northern Alps and the Southern Alps. This factor is combined with a complex topography formed by Tertiary tectonic activity followed by glaciations. Elevation ranges from areas below 100 m up to Mont Blanc culminating at 4810 m. A secondary continentality gradient runs from Atlantic climatic influence on western external Alps to continental conditions in the internal Alps. This W-E gradient is also coupled with a geological gradient from sedimentary substrates in the external Alps to crystalline substrates in the internal Alps. Overall, it is interesting to note a strong and fine-scaled heterogeneity of the indicator of land use intensity (hemeroby) across the French Alps, though with a clear W-E gradient and a weaker N-S gradient of land use intensity.

Piedmonts and longitudinal valleys in the western part of the study area concentrate much of the French Alps arable lands, which are generally dedicated to cropping or mixed farming. Thanks to more favourable conditions (gentle slopes, smoother climatic conditions), many land uses tend to concentrate in limited space, leading to a high rate of intensity in human practices (i.e. high hemeroby) and thus leaving very little space (if any) for natural areas. High-density urban areas in the valleys, where the labour market is concentrated, are surrounded by residential municipalities from where people usually commute to the cities every day while enjoying the pleasant surroundings. This results in a high fragmentation of the territory and an important pressure on natural habitats and ecosystem functions as well as on ecological connectivity.

In the areas of intermediate altitude of northwestern Alps, land use appears more diverse and associates forested areas with arable lands, grasslands and pastures. A large part of the territory is covered by standard alpine regions, characterized by a modest decline of agriculture and a balance between migration and birth rates that prevents over-ageing. Forms of tourism are contrasted as some specific areas concentrate highly impacting activities, notably during winter time, while the rest of the territory is concerned by an overall quite low touristic intensity.

Due to natural constraints (altitude, climate, slope), part of the northeastern French Alps has been dedicated to extensive livestock farming that maintain landscapes open with pastures and grasslands. Agriculture in this part of the massif remains dynamic although patches of forgotten rural areas undergoing abandonment are also present. This trend of agricultural abandonment is partly responsible for the overall low hemeroby of this sub-region, together with the large forested and open areas where impacts from human activities and settlement are lessened by physical constraints and distance to attractive centres. However, this sub-region also comprises dynamic rural areas, characterised both by a rural location and a dynamic labour market, and rural retreats where good transport links allow city workers to live in remote hinterlands. The sub-region additionally experiences a particularly positive development of tourism, mainly during winter time, with corresponding impacts on high altitude sensitive areas through infrastructure development. These complementary features lead to a highly diverse and attractive cultural landscape, although undergoing modification due to land use changes.

High altitude areas of the internal Southern Alps present a contrasted image as their economy is much less dynamic than in the North. Extensive agricultural activities characteristic of this sub-region represent an important opportunity for local employment. Tourism is mainly rural and small scaled. However, the steepest and most constrained areas (e.g. highly erodible soils) undergo a significant decline in farming activities and also in population since World War II. This results in the closing of traditional landscapes by natural afforestation.

At lower altitude, in the South, more gentle natural conditions are suitable for cropping or mixed farming, in addition to extensive livestock farming. Overall, this sub-region typically includes rural areas with low tourism intensity, poor transport infrastructures and an ageing population. The combination of agricultural lands with large areas covered by forests or semi-natural habitats results in a rich traditional landscape, although undergoing modification due to the same significant trend of agricultural abandonment than in the internal Southern Alps. Overall, human impacts on ecosystems remain moderate as management intensity overall decreases with agricultural changes, although local contrasts can appear with areas undergoing an intensification of agricultural practises at the same time.

C. The Alps from a governance perspective

Governance at the scale of the whole massif is coordinated by an international treaty, the *Alpine Convention*, which “seeks to protect the natural environment and cultural integrity of the Alps while promoting the region’s development” (http://www.alpconv.org). This Convention concerns the eight States over which the massif expands (Austria, Germany, France, Italy, Liechtenstein, Monaco, Slovenia and Switzerland) as well as the European Union. Eight *Protocols* contain the specific measures implementing the principles laid down in the framework Convention. They propose “concrete steps to be taken for the protection and sustainable development of the Alps" (http://www.alpconv.org) regarding i) spatial planning and sustainable development, ii) conservation of nature and countryside, iii) mountain farming, iv) mountain forests, v) tourism, vi) energy, vii) soil conservation and viii) transport. While the Alpine Convention framework was opened to signature in 1991 and entered into force in 1995, the process of ratification of protocols is slower. All member states agreed on the protocols in 2002 and are since then ratifying them. This step aims at translating protocol objectives into national legislations which alone have full legal effects and actually bound the States to implement the protocol. If France already ratified all protocols, some countries still need to further advance in their integration of the Convention objectives at national scale.

In France, the massif is also recognised *per se* in governance through the Massif Committee ('*Comité de massif*'). This Committee is a consultative organisation concerned by the planning, development and conservation of the massif at national scale. It has a role of counsel and coordination among the administrative levels of NUTS-2 and -3 levels encompassed in its perimeter. Different framework documents are proposed to assess the state of the French Alps and to plan their sustainable future (e.g. Massif Interregional Planning and Management Scheme – Interregional Operational Program for the Alpine Massif). Lower scale policy documents need to account for these broad objectives in their specific declinations.

D. Preliminary conclusions

The French Alps are characterised by contrasted social and ecological features, spatially constrained by a complex mountain abiotic setting. Various uses are made of ecosystems, with at least agriculture, forestry and tourism exerting a significant influence on ES and landscapes. Combined and increasing impacts from land use and climate changes are increasingly putting under pressure its (semi-)natural areas of overall high sensitivity, making their management even more challenging. Thus, the assessment of bundles of ecological parameters (i.e. both ES and biodiversity variables) over the French Alps appears critical as, in addition to this region's specific biophysical conditions, it hosts high levels of diversity in terms of species, cultural landscapes and human uses. The administrative organisation of the French massif encompasses multiple nested levels which are sometimes overlapping (Alpine convention perimeter vs. regions and départements). Their joint influence through policy shapes land allocation and management, with subsequent impacts on ES and biodiversity, together with social dynamics. Overall, a better understanding of the various components and relationships within the social-ecological system is needed to support future management and governance of natural resource issues over the French Alps (Stevenson 2011).

IV. Research questions and structure of the manuscript

Past years have witnessed a convergence of conceptual frameworks across disciplines and spheres (academic / management / policy) (Stevenson 2011), leading ES scientists to explicitly target the exploration of social-ecological systems as a research priority (Anton et al. 2010). And yet, few assessments actually explore with equal intensity the ecological and social systems and further interrelate their findings to propose an integrative understanding of the system (Nicholson et al. 2009, Chan et al. 2012, Martín-López et al. 2014). Moreover, the generality and strengths of associations between ES and biodiversity still need to be substantiated (Balvanera et al. 2013). Overall, the assessment of social-ecological systems integrating multiple value-domains and the identification of bundles of ES and biodiversity parameters appears a promising and yet under-explored option.

Additionally, in the French Alps, complex changes at global scale (climate) and local scale (land use changes – management modality changes – societal changes) alter cultural landscapes and put under pressure sensitive alpine ecosystems and species. Overall the French Alps face increased tensions over ES supply due to an increased land fragmentation from urban sprawling and the multiplicity of demands from various stakeholders, which raise issues of land allocation and management at nested scales. Consequently, there is need to deepen our understanding of the determinants and consequences of ES management in a 'social-ecological perspective'.

The overarching objective of my PhD is thus to approach trade-offs and synergies among ES and biodiversity in the social-ecological system of the French Alps through a multi-layered assessment mobilising biophysical and socio-cultural value-domains.

To progress in this endeavour, I addressed the four following questions, each developed specifically in one Chapter of this manuscript:

1) **What are the spatial patterns and determinants of ES and biodiversity co-variation, regarding their biophysical values?**
   - ⇨ Chapter I presents a quantitative biophysical assessment of interactions between ecosystem services and biodiversity. After compiling maps for 16 ecosystem services and two biodiversity parameters at a 1 km² resolution for the entire French Alps, spatial patterns of trade-offs and synergies were explored using a series of statistical analyses of increasing complexity. Results were structured to provide insights for sound environmental governance at multiple scales. This assessment was submitted as a paper in *Journal of Applied Ecology* which is currently pending minor revisions.

2) **How do ES, biodiversity and external variables interact in complex social-ecological systems?**
   - ⇨ Chapter II addresses the need for an increased understanding of influence relationships within the social-ecological system. We proposed an innovative theoretical framework that makes explicit the relationships among ES facets, biodiversity and external variables. To test the operational potential of this framework, we carried out a consultative process with stakeholders of regional expertise to inform our description of the alpine system. Our framework appeared relevant to communicate on environmental management and to foster dialogue and social learning among diverse stakeholders. This work will be submitted as a paper in *Ecology & Society* within the next few weeks.

3) **How effective is the alpine policy mix at enhancing biodiversity and ES in the specific context of interactions among agriculture, tourism and biodiversity?**
   - ⇨ Chapter III focuses on governance and on the effectiveness of policy instruments for sustaining ES supply and conserving biodiversity. In the context of the CONNECT project, we tested a methodology developed by our partners to assess the environmental effectiveness of a policy mix. We thoroughly assessed 10 policy instruments currently used to regulate influence relationships at the interface between biodiversity, agriculture and outdoor tourism. In addition to

classical policy mix criteria, we paid particular attention to the rebound effects of these policies, i.e. their positive and negative effects on untargeted environmental aspects. The policy mix assessment was addressed by an extensive literature review and further comforted by individual interviews. A policy brief was designed to communicate on our findings with stakeholders at regional level. I supervised the Master student in charge of this assessment together with Sandra Lavorel. Publication of the results is planned within the coming months.

4) **How do scientists in environmental science relate their work with society and governance?**
   - ⇨ Chapter IV is conceived as a personal exploration of the conceptual and ethical issues linked to research in the ES domain. It addresses the interrogations I faced while discovering this concept and related controversies, as well as the questions I sought to answer regarding roles of scientists in society. I explored an interdisciplinary literature from ecological, economical and philosophical backgrounds and aimed at interweaving their insights to characterise, in the current academic and social setting, the postures adopted by environmental scientists in general, and in my work in particular.

A general discussion complements these chapters and highlights cross-cutting issues addressed throughout my work. Two additional papers where I participated as co-author are also included and available in the Appendix (in this manuscript, all pages integrating the papers can be distinguished by an additional black border).

Figure 4 summarises the different relationships among the concepts I mobilised for this study and relates them to the chapters where they are specifically explored.

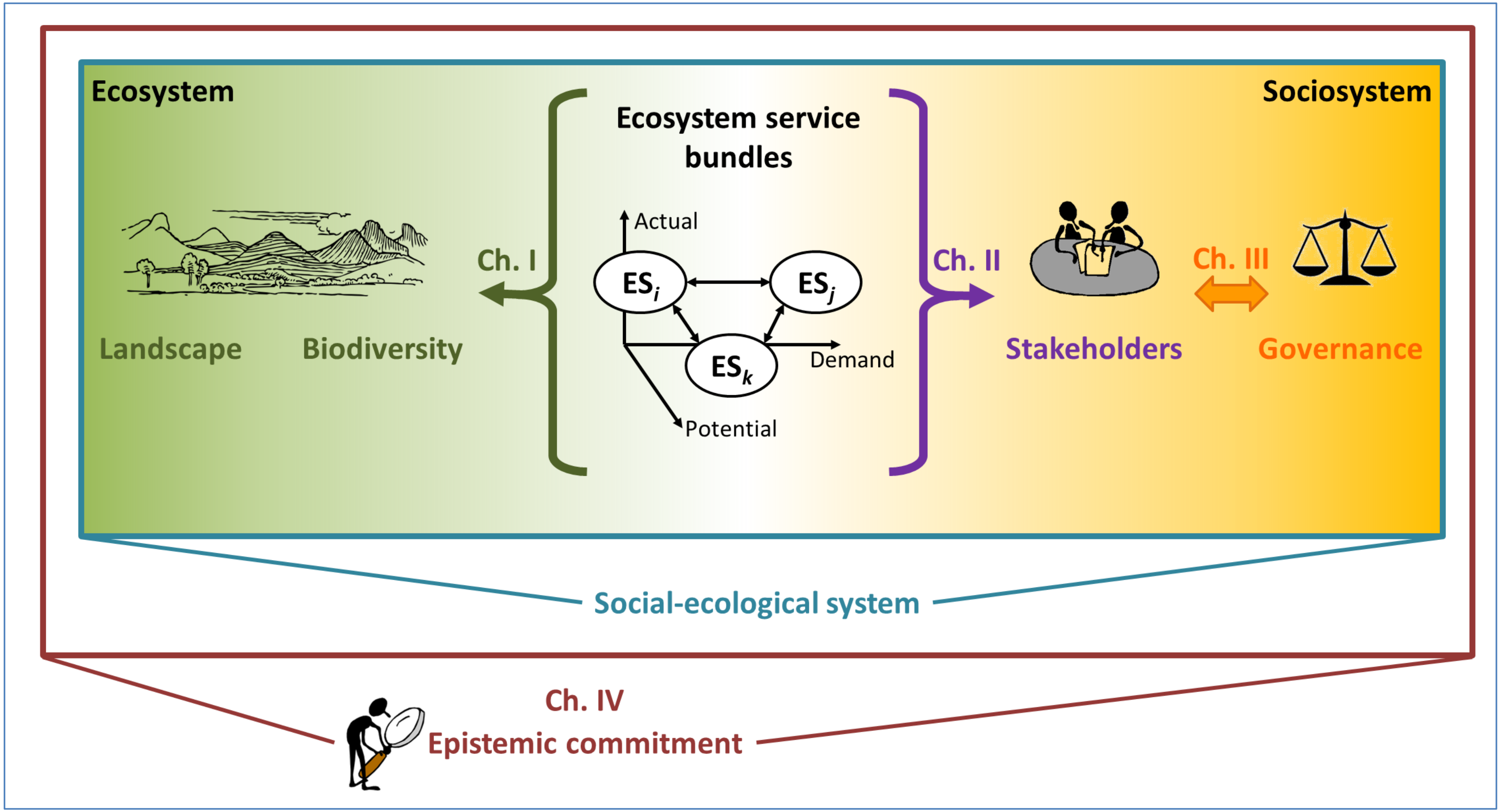

Figure 7: Synthetic scheme of the relationships among the main concepts mobilised in my PhD and related Chapters.

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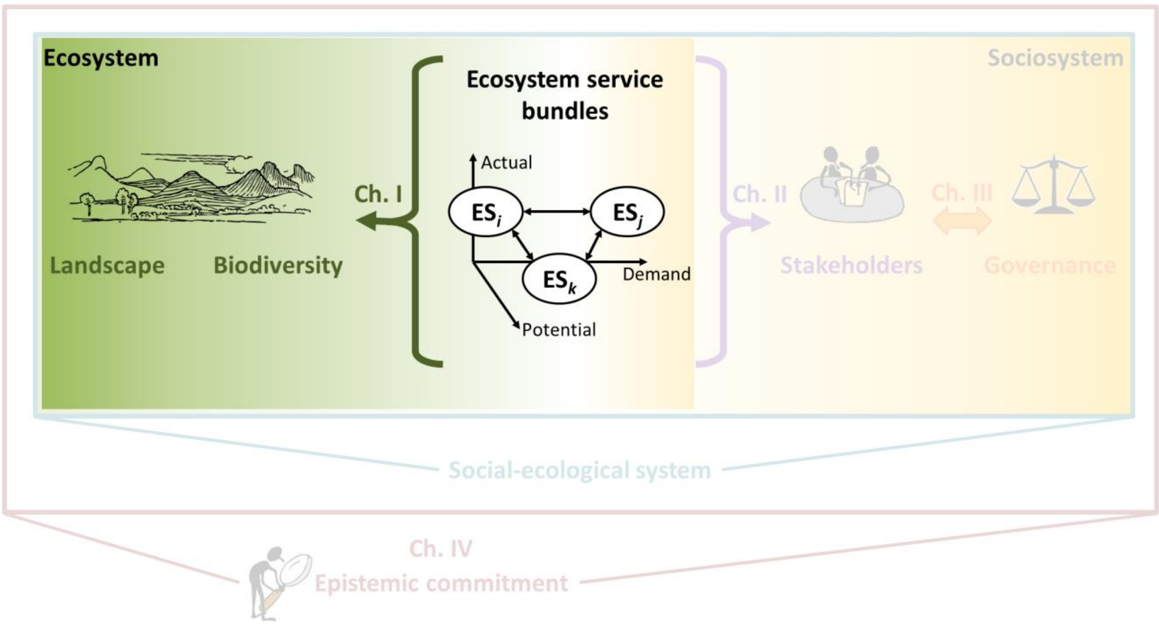
Chapter 1 - Biophysical assessment of ecological parameters bundles



Chapter 1 - Biophysical assessment of ecological parameters bundles

Chapter I aims at exploring bundles of ecological parameters (EP), i.e. biodiversity and ecosystem services, using a biophysical perspective.

Chapter I is structured in six sections:

- Section I presents the **specific research questions** related to our biophysical perspective on EP bundles.
- Section II introduces the **dataset**: an unprecented array of 16 ES and 2 biodiversity parameters for the French Alps.
- Section III comments our **methodological choices and issues** for EP modeling and mapping.
  Section IV briefly summarizes the objectives and characteristics of the **statistical analyses** we performed to explore EP bundles.
- Section V is a **paper**, submitted to the *Journal of Applied Ecology*, that incorporates a presentation and discussion of our **main results** (pages highlighted by a black border).
- Section VI concludes by a **synthesis** of main insights and issues from this biophysical assessment, and highlights their relevance for **governance of natural resources**.

I. Specific research questions

The overarching objective of this first chapter is to explore how biophysical values of ES and biodiversity parameters co-occur spatially over the French Alps, and to relate their synergy and trade-off patterns to broad landscape features. This objective was approached through the four following questions:

1) What are the spatial distributions of individual ecological parameters relevant for the French Alps?
2) Which bundles of ecological parameters can be identified at various scales?
3) How do ecological parameters relate to landscape features?
4) Are mosaic landscapes more multifunctional than homogeneous ones?

To answer those questions, a series of statistical analyses were performed on a set of 18 ecological parameters selected regarding their relevance for ecosystem and natural resource management in the French Alps.

Additionally, we were concerned by the potential of our analysis and related findings for supporting the governance of natural resources. Thus, we thoroughly explored how our results could provide a sound basis for existing governance instruments or alternatively could provide interesting insights for ecological relations seldom targeted to date.

II. Introduction to the ES and biodiversity dataset

We used an unprecented array of 18 ecological parameters composed of 16 ecosystem services (ES) and two biodiversity parameters.

A. Selection of the ecological parameter set

For the assessment of ecological parameters bundles, we chose which variables would be represented. Although required, this choice holds an overarching influence on scientific conclusions and also on their communication to stakeholders. This particularly holds true when the characterisation of ES and biodiversity is to be used for land management or land planning. Regarding biophysical assessment for the French Alps, justification of the ecological parameters selected is twofold, in relation to alpine context knowledge as well as to data and model availability.

First, our choice was grounded on knowledge of the alpine context. Indeed, the core set of ES was proposed by the scientific team based upon previous project experiences (VISTA, VITAL, VOLANTE…). Additional inputs arose from local stakeholders who shared their concerns and priorities with us during informal discussions. For instance, leisure hunting was added due to the complex stakeholder interplay that was described around this ES (including forest managers, hunters and tourists) and that affected indirectly the biophysical ability of ecosystems to supply other ES such as wood production.

Second, the final set of ES reflects data and model availability. As noted by Eigenbrod et al. 2010, “Perhaps the greatest obstacle to substantial progress in assessing ecosystem services is a lack of data – there is simply none available for most services in most of the world.” We faced the same issue in the French Alps assessment. For instance, lack of existing spatial data on wood energy volumes harvested forced us to keep an aggregated wood production variable. We initially wanted to use of two complementary variables describing on the one hand industrial and lumber wood production and on the other wood energy production. The same lack of spatial data was faced regarding biodiversity variables: invertebrate ecological ranges and abundances are still unexplored to the point of obtaining their spatial distributions at the French Alps scale, despite their uncontested interest per se as well as basis of ES supply. In addition to this general lack of data for some EP, we faced a lack of consistency in available data across the entire study area. This concerned either spatial factors, in relation to the administrative distinction between Rhône-Alpes and Provence-Alpes-Côte d’Azur regions (e.g. for hydro-energy datasets), or species-related factors. As an example for the last point, leisure hunting was considered under its actual supply facet (i.e. actual total number of wild ungulates killed during one hunting season) as the potential supply facet (i.e. population size of game species) was available for some species, as for red deer, but not all, as for wild boar and despite their huge numbers hunted each year. Finally, we used preferentially readily available and user-friendly models due to time constraints. As a result, we did not explore the regulation ES of maintenance of air quality, which could have been interestingly added to our dataset, but for which we lacked experience, competent collaborators and easy-to-use models. However, more time would have allowed us to overcome those limitations and could be considered in subsequent ES biophysical assessments, as by using the i-Tree software (https://www.itreetools.org/), which is based on the structure of tree communities to quantify the ES they supply, including biotic contribution to the maintenance of air quality.

Overall, and despite technical constraints, we contend that our set of 18 ecological parameters remains highly informative for natural resources management over the French Alps and that it covers most relevant features from ecological and social points of view.

B. Description of ecological parameters

Below we present briefly the set of 18 ecological parameters used for the biophysical assessment. Parameters are displayed by main category: provisioning ES (Table 1), cultural ES (Table 2), regulating ES (Table 3) and biodiversity parameters (Table 4). My inputs in the process of data collection, modeling and mapping are specified for each variable.

Further details on ecological parameters are to be found in the paper presented in section V of this chapter (Supporting Information S1.A). There, we provide elements for descriptions of ecological parameters standardised as proposed by Crossman et al. 2013, with additional information on methods and data sources following Martínez-Harms & Balvanera 2011.

Table 1: Short description of the four provisioning ES used in the biophysical assessment of ecological parameters over the French Alps.

Ecological parameter	Variable (unit)	Short description	My inputs
Agricultural production	Yields (kg/km²/yr)	**Aggregation of yields** for annual crops, vineyards and orchards for 2009.	Data collection Method building Mapping (collaboration with C. Byczek - LECA)
Forage production	Yields (kg dry matter /km²/yr)	**Aggregation of yields** of pastures, meadows and mountain grasslands, defined at the level of the "département" for 2009. Yields for each kind of pasture, meadow or mountain grassland were refined according to the likelihood of their presence at a certain altitudinal range in a given eco-region.	Data collection Method building Mapping (collaboration with C. Byczek - LECA)
Wood production	Harvestable potential from woody biomass (Gg dry matter /km²/yr)	**Potential woody biomass supply** estimated for 2010 for stemwood and logging residues. Theoretical biomass potential was estimated from forest inventory data using EFISCEN model and corresponds to bio-physical potentials of the forests. Social, technical and environmental constraints reducing the availability of woody biomass were quantified and combined to theoretical potentials to assess the realisable potential.	- (collaboration with VOLANTE project)
Hydro-energy potential	Theoretical total potential hydroelectric power (classes)	**Theoretical total potential** for hydro-energy production by river basin (mean area of 135km²), according to physical assets of the territory (e.g. slope, rivers length and flow). Biophysical characteristics of the basin impact hydro-energy potential by modulating the amount of rainfalls and the runoff volumes, as well as the uptakes by vegetation cover. Hydro-energy potentials were discretised into 5 classes using French Water Agency thresholds.	Data mining

Table 2: Short description of the five cultural ES used in the biophysical assessment of ecological parameters over the French Alps.

Ecological parameter	Variable	Short description	My inputs
Recreation potential	Recreation potential (adimensional index)	**Potential for daily recreation** provided by ecosystems, in relation to the presence of certain ecosystems (i.e. forest, coastline), certain ecosystem characteristics (i.e. naturalness) and their accessibility.	- (collaboration with VOLANTE project)
Tourism	Territorial capital of rural tourism (adimensional index)	**Potential for 'rural tourism'** incorporating the supply of 'beach tourism,' of attractions for winter tourism, of attractions for nature tourism and assets of symbolic capital.	- (collaboration with VOLANTE project)
Leisure hunting	Density of wild ungulates killed (number of animals killed / km² / yr)	**Number of wild ungulates killed** per year (red deer, chamois, Corsican and Mediterranean mouflon, roe deer and wild boar). This definition includes the ability of ecosystems to host biodiversity, and the demand society makes for game. All species are given an equal weight; we do not consider possible hunters' preferences for one or the other species.	Data collection (collaboration with ONCFS / FDC / FNC) Method building Mapping
Protected plant species	Species richness (number of species/km²)	**Overlay of potential ecological niche distributions** for 45 protected plant species hosted by the French Alps. Protected species are the ones concerned by IUCN French Red List status critical, endangered and vulnerable.	- (collaboration with W. Thuiller - LECA)
Protected vertebrate species	Species richness (number of species/km²)	**Overlay of potential ecological niche distributions** for 107 protected vertebrate species hosted by the French Alps. Protected species are the ones concerned by IUCN French Red List status critical, endangered and vulnerable.	- (collaboration with L. Maiorano - Università di Roma "La Sapienza")

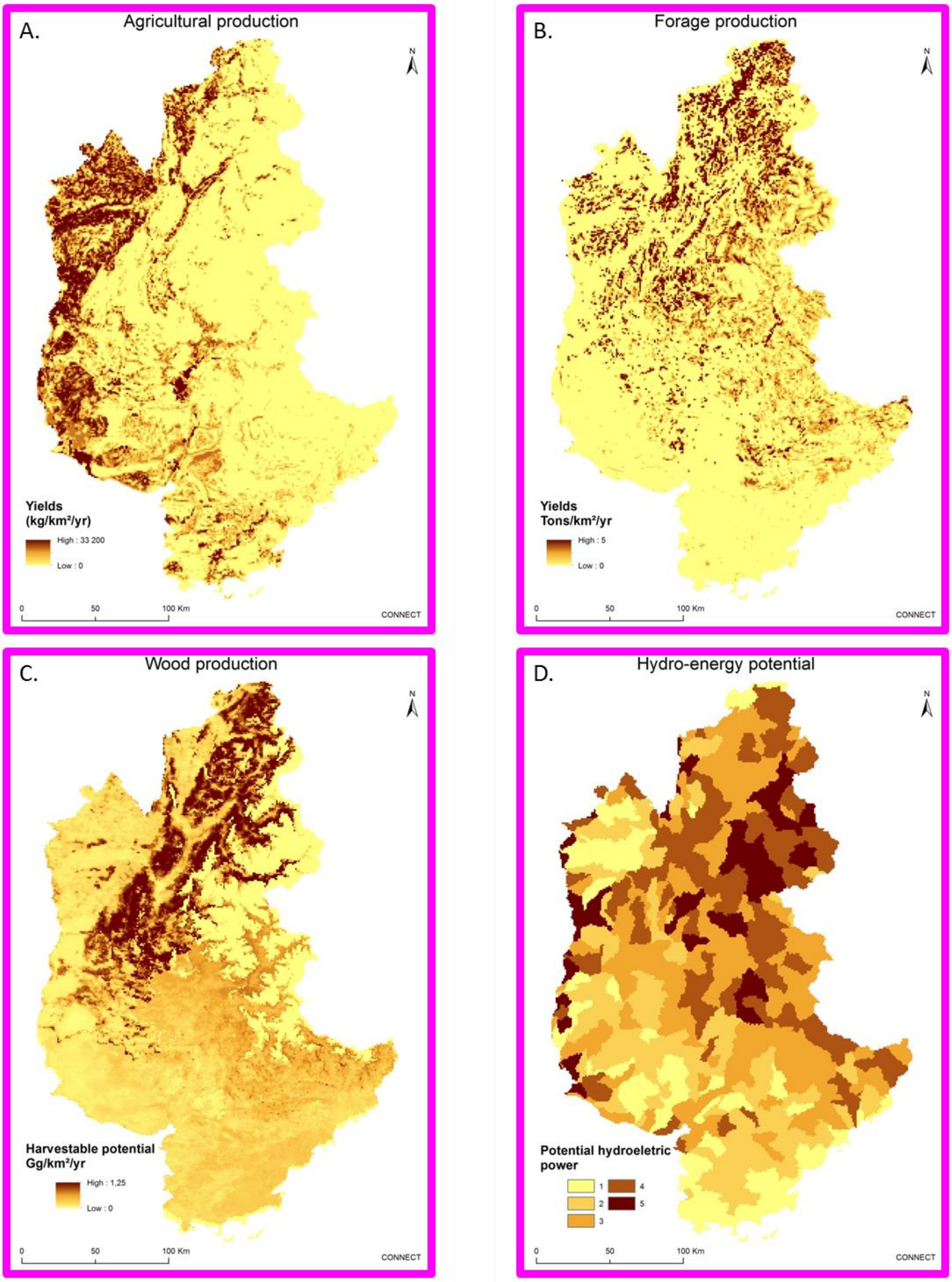
Table 3: Short description of the seven regulating ES used in the biophysical assessment of ecological parameters over the French Alps.

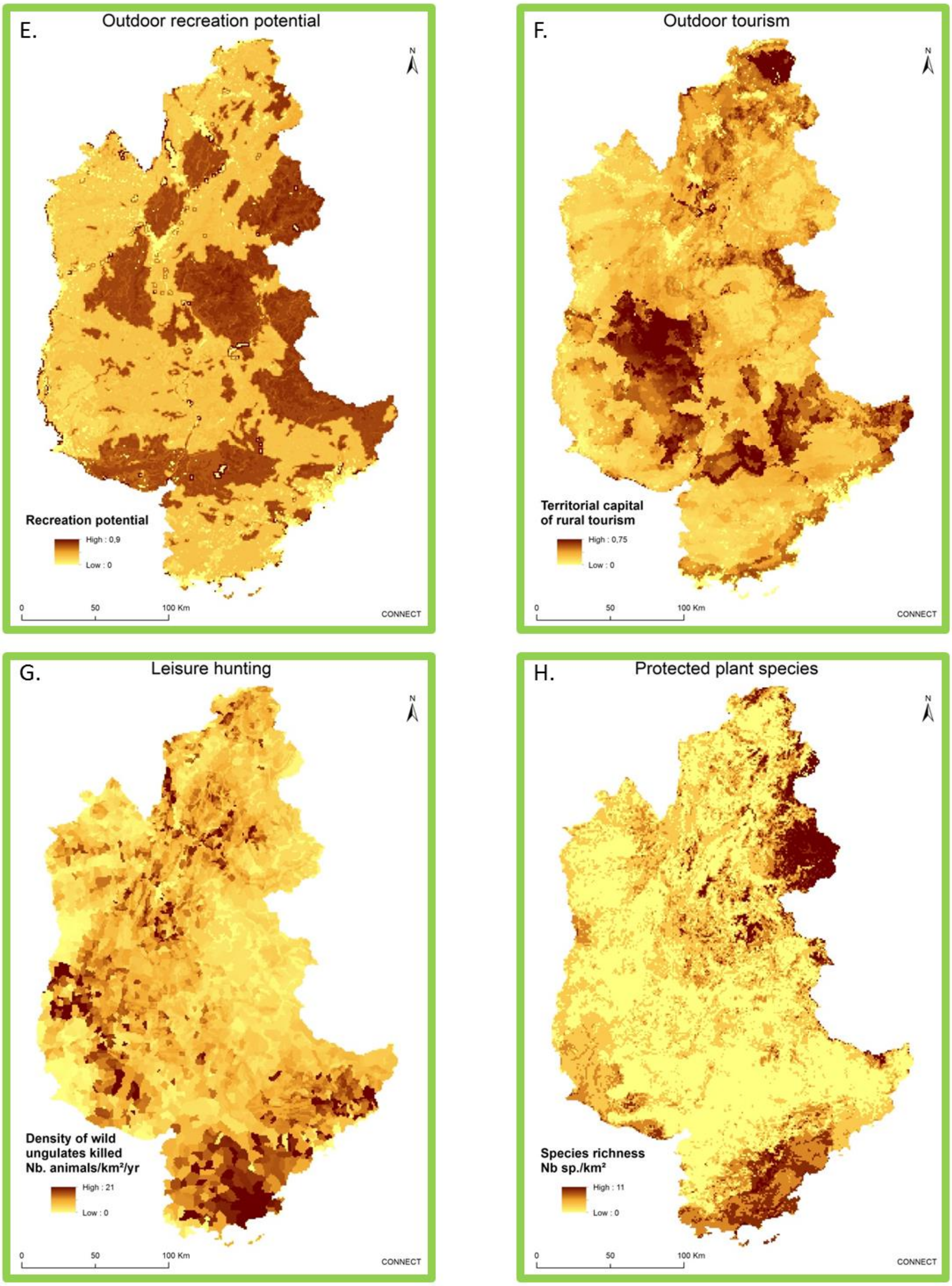
Ecological parameter	Variable	Short description	My inputs
Erosion mitigation	Biotic contribution to erosion risk mitigation (classes)	**Ability of biotic factors to make erosion risk decrease.** Classes represent the difference between potential risk class (ignoring vegetation role) and effective risk class (including vegetation role). Potential and effective risks were determined using the empirical model RUSLE adapted to the Alps conditions.	Data mining (collaboration with ClimChAlp project) Method building Mapping (collaboration with C. Byczek - LECA)
Protection against rockfalls	Potential to protect against gravitational hazards (adimensional index)	**Ability of forests to decrease rockfall hazard** i.e. presence of forests susceptible of intercepting or slowing rocky projectiles between probable starting points and actual sensitive areas linked to human infrastructures and presence. Specific forestry model RockForLIN and computer utility RollFree were used.	- (collaboration with F. Berger - IRSTEA)
Chemical water quality regulation	Nitrogen retention capacity (tN/km/year)	**Amount of nitrogen retained in water bodies** (proportion of potential input). The model considers the input of diffuse and point sources of total nitrogen and estimates the nitrogen fraction retained during the transport from land to surface water (basin retention) and the nitrogen fraction retained in the river segment (river retention). The statistical proxy modeling uses GREEN model.	- (collaboration with VOLANTE project)
Physical water quantity regulation	Relative water retention in relation to flood regulation (adimensional index)	**Landscape capacity to modify the river discharge** after heavy precipitation events potentially causing flood events. This index is based on the variability of the peak discharge at the outlet of a catchment in dependence of land use and soil distribution. We used the model STREAM, a conceptual empirical hydrological model by the Institute for Environmental Studies of the Vrije Universiteit Amsterdam (IVM-VU).	- (collaboration with VOLANTE project)
Biological control of pests	Natural predator species richness (number of species/km²)	**Richness in species providing natural control** of invertebrate and rodent pests. It was obtained through the overlay of potential ecological niche distributions for 110 vertebrate species considered as natural predators of agricultural pests.	- (collaboration with VOLANTE project)
Pollination	Relative landscape suitability for pollinators (adimensional index)	**Relative capacity of ecosystems to support crop pollination.** This index relates to the availability of floral resources, bee flight ranges and the availability of nesting sites.	- (collaboration with VOLANTE project)
Carbon storage	Amount of carbon stocks (tC/km²)	**Amount of carbon stocked** in above-ground, below-ground biomass, dead organic matter and soils. We used the InVEST platform, module Carbon, and considered specifically stocks in forests, grasslands and agricultural areas.	Data collection Method building Mapping (collaboration with C. Byczek - LECA)

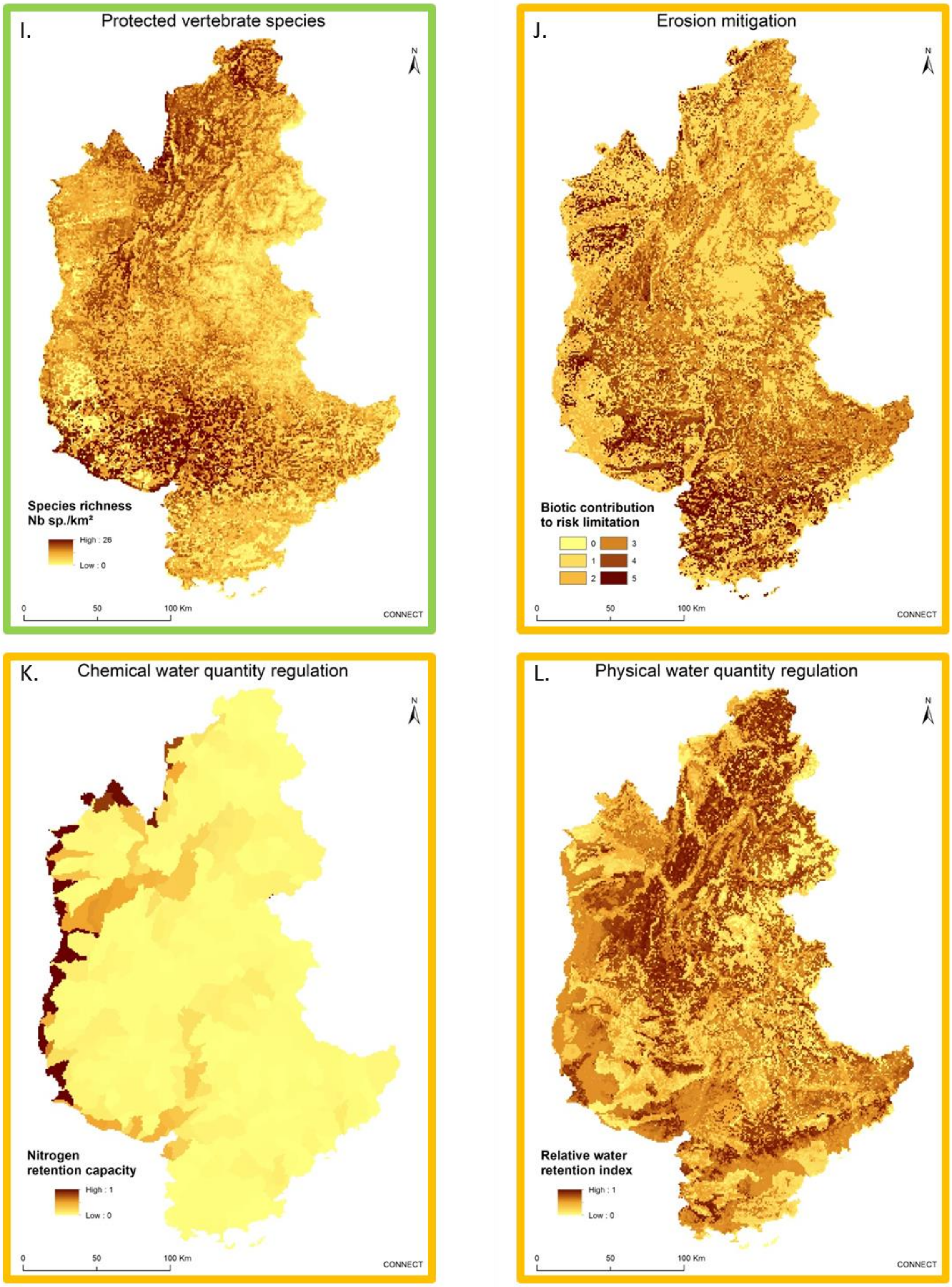
Table 4: Short description of the two biodiversity parameters used in the biophysical assessment of ecological parameters over the French Alps.

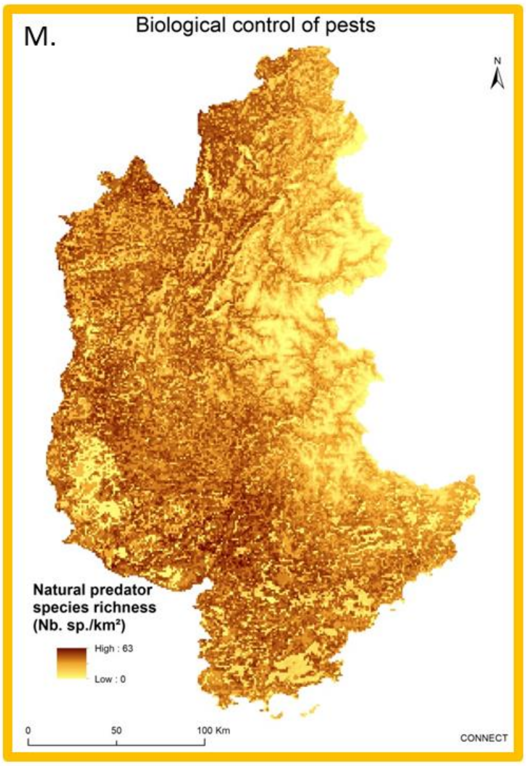
Ecological parameter	Variable	Short description	My inputs
Plant diversity	Species richness (number of species/km²)	**Overlay of potential ecological niche distributions** refined with presence data and habitat preferences for 2748 plant species hosted by the French Alps. Primary field data were used to model ecological niche distributions based on biophysical information.	- (collaboration with W. Thuiller - LECA)
Vertebrate diversity	Species richness (number of species/km²)	**Overlay of potential ecological niche distributions** refined with presence data and habitat preferences for 380 vertebrate species hosted by the French Alps. For each species, a suitability score was assigned by experts and literature to land cover classes to distinguish land-use/land-cover classes that represent suitable from inadequate habitats. Elevation range where each species can be found and maximum distance to water were combined with habitat suitability scores to refine the available extents of occurrence, as well as all freely available presence points.	- (collaboration with L. Maiorano - Università di Roma "La Sapienza")

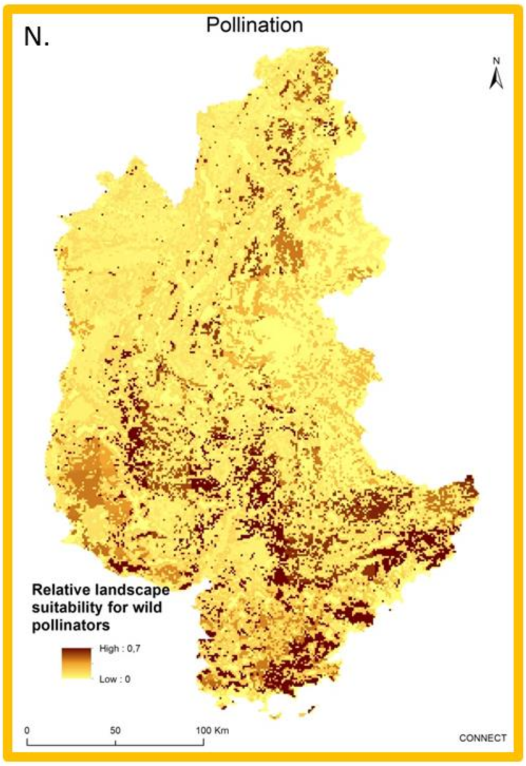
Below we propose individual maps of the parameters, except for the regulating ES 'Protection against rockfalls' that is not displayed due to data confidentiality commitments (Figure 1).

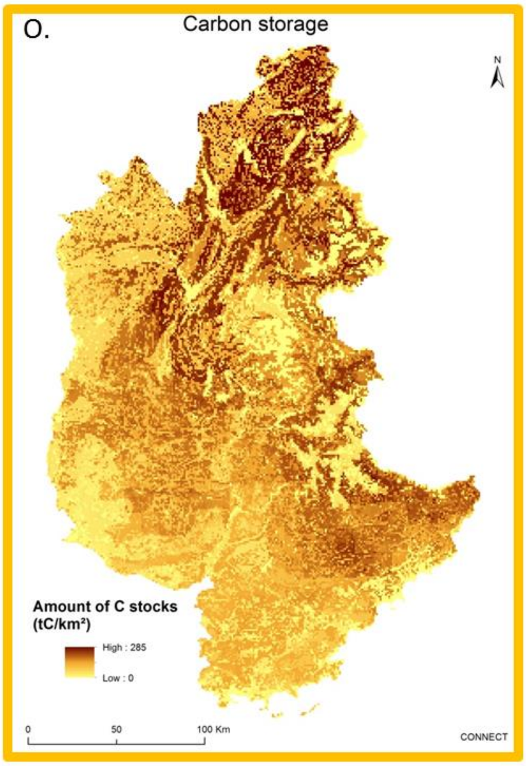



E.
Outdoor recreation potential
Recreation potential
High : 0,9
Low : 0
0
50
100 Km
CONNECT
F.
Outdoor tourism
Territorial capital
of rural tourism
High : 0,75
Low : 0
0
50
100 Km
CONNECT
G.
Leisure hunting
Density of wild
ungulates killed
Nb. animals/km²/yr
High : 21
Low : 0
0
50
100 Km
CONNECT
H.
Protected plant species
Species richness
Nb sp./km²
High : 11
Low : 0
0
50
100 Km
CONNECT


I.
Protected vertebrate species
Species richness
Nb sp./km²
High : 26
Low : 0
0
50
100 Km
CONNECT
J.
Erosion mitigation
Biotic contribution
to risk limitation
0
1
2
3
4
5
0
50
100 Km
CONNECT
K.
Chemical water quantity regulation
Nitrogen
retention capacity
High : 1
Low : 0
0
50
100 Km
CONNECT
L.
Physical water quantity regulation
Relative water
retention index
High : 1
Low : 0
0
50
100 Km
CONNECT


M.
Biological control of pests
N
Natural predator species richness (Nb. sp./km²)
High : 63
Low : 0
0
50
100 Km
CONNECT


N.
Pollination
N
Relative landscape suitability for wild pollinators
High : 0,7
Low : 0
0
50
100 Km
CONNECT


O.
Carbon storage
N
Amount of C stocks (tC/km²)
High : 285
Low : 0
0
50
100 Km
CONNECT

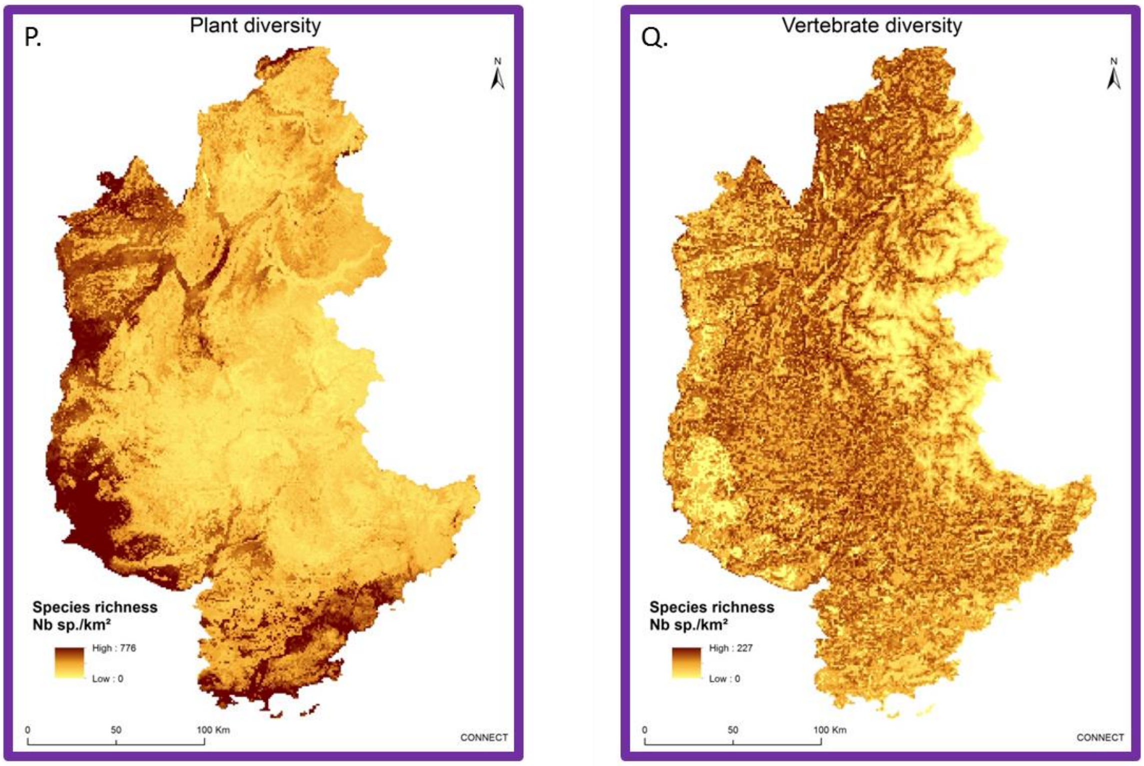


Figure 1: Resulting maps for the ecological parameters modelled and analysed: provisioning ES (pink border, Fig. 1.A. to 1.D.), cultural ES (green border, Fig. 1.E to 1.I), regulating ES (orange border, Fig. 1.J to 1.O.) and biodiversity variables (purple border, Fig. 1.P and 1.Q).

C. Concerning the cultural value of biodiversity

A short aside is presented here to discuss our choice for representing the cultural value of biodiversity. The two ES explored were linked to ecosystem richness in protected vegetal and vertebrate species, according IUCN French Red List (status critical, endangered and vulnerable). However, our initial thought was to use a restricted list of 20 'iconic' species, as selected by stakeholders of regional expertise. The objective would have been to specifically focus on the particular cultural value attributed to the presence of certain species in this area. Patrimonial species are linked to specific territories which are responsible for their conservation (as species distribution is greatly encompassed within them) and whose cultural identity partly relies on their presence. Iconic species are not always protected by a specific legislative status but can be seen as iconic species for ecosystems and their functioning in given areas. In France, the legislative Strategy for the Creation of new Protected Areas (SCAP - http://scap.espaces-naturels.fr/) explicitly based the justification of new protective perimeters on the actual presence of such species, selected at regional level from a national list.

We consulted twelve stakeholders from the academic sector and from official structures in charge of nature conservation. We proposed them to pick up from the list of iconic species for which spatial distributions were available the 10 plant species and the 10 vertebrate species that seemed of prominent interest of conservation to them. The top-ten species most cited are proposed in Table 5, and main determinants for their selection discussed below (complete information on this consultation and outputs is to be found at the end of the manuscript in the Appendices from Chapter I (Section B) – in French).

Table 5: Most cited species for the selection of alpine iconic species of great conservation interest.

Vertebrates		Plants	
Latin name	Common name	*Latin name*	Common name
Vipera ursinii	Meadow viper, Ursini's viper	*Eryngium alpinum L.*	Alpine sea holly, Alpine eryngo, Queen of the Alps
Lynx lynx	Eurasian lynx	*Astragalus alopecurus Pall.*	-
Lutra lutra	European otter	*Dracocephalum austriacum L.*	-
Rhinolophus hipposideros	Lesser horseshoe bat	*Cypripedium calceolus L.*	-
Speleomantes strinatii	Cave salamander	*Juniperus thurifera L.*	Spanish Juniper
Lepus timidus	Mountain hare	*Liparis loeselii (L.) Rich.*	-
Gypaetus barbatus	Bearded vulture	*Aquilegia alpina L.*	Alpine Columbine
Hieraaetus fasciatus	Bonelli's eagle	*Potentilla delphinensis Gren. & Godr.*	-
Tetrao tetrix	Black grouse	*Saxifraga florulenta Moretti*	-
Aegolius funereus	Boreal owl	*Serratula lycopifolia (Vill.) A.Kern.*	-
		Marsilea quadrifolia L.	Four Leaf Clover

Most frequent justifications for the selection of iconic vertebrate species were three-fold: i) species of small population sizes that should be supported by conservative measures to be maintained, ii) species considered as umbrella species which conservation could benefit to many associated others, and iii) species with important functional roles, as predators and scavengers. The three most-cited arguments for plant species selection differed: i) species considered as flagships for the French Alps area, ii) species valued for their aesthetic quality, and iii) species with current status of protection that already demonstrates their need to be protected. It is interesting to note the distinct nature of determinants for species selection. Indeed, vertebrates were selected accordingly to scientific criteria (abundance and trophic characteristics) while subjective criteria were mobilised for plant species (flagship and aesthetic species).

Our objective with this restricted list of species was to include a cultural dimension to biodiversity variables in environmental assessments. However, we faced a low response rate from stakeholders we solicited, with only ten usable short lists of species when we aimed at twice and with marked oppositions to answering us from some nature conservation organisations. Two hypotheses can explain this failure. First, we did not anticipate the political weight given to this selection, that we regarded only as an academic focus on iconic species ‘of special conservation interest’. Some stakeholders contested the relevance of focusing on 20 species to represent the cultural value of biodiversity in particular because they feared inappropriate uses of such ‘stakeholder approved’ lists for designing conservation strategies. Second, we proposed to pick up species from the official list of regional iconic species (SCAP), but several respondents were reluctant to start from this list as they questioned its consistency and relevance.

Thus, we decided not to use the short list of iconic species as it appeared too subjective and of low reliability regarding the restricted number of respondents. We finally focused on existing official lists of species with need of conservation, and chose to represent species selected by the IUCN French Red List status critical, endangered and vulnerable. After carrying out the process of analysis and having presented results to various audiences, this choice appears relevant as understanding of the proxy for iconic species was straightforward and legitimacy of Red List species unquestioned.

III. Modeling choices and issues

We discussed in the previous section the importance and determinants of ecological parameter selection. Here, we expand on modeling and mapping issues faced for the biophysical assessment of ES and biodiversity bundles.

Indeed, the process of representing natural capital and processes is challenged by the inherent complexity of nature. In particular, ES are the expressed consequence of multiple interacting and often nonlinear ecological processes (Briner et al. 2013) and furthermore vary depending on human land allocation and management choices (Lavorel et al. 2011, Maskell et al. 2013). Such complexity cannot be captured fully by ecological models, leading to limitations in the range of ecological processes considered and to the use of proxies (Eigenbrod et al. 2010, Seppelt et al. 2011). Both proxy use and modeling assumptions distort the reality and reinforce the importance of the choices made to determine through which prism the ES is explored. Finally, any interpretation of ES mapping and bundling requires in-depth understanding of those modeling choices.

A. Balancing model complexity and informativeness

Many studies have been carried out to explore trade-offs and synergies between restricted sets of ES (e.g. Egoh et al. 2008, Garcia-Nieto et al. 2013), and their co-variation with biodiversity (e.g. Chan et al. 2006, Bai et al. 2011). They enabled an in-depth understanding of the relations between variables explored but calls have been made to widen the range of ES considered, by including more cultural and social aspects (Chan et al. 2012) and by considering numerous ES at the same time (up to 29 in Burkhard et al. 2009). Our assessment over the French Alps sought to expand in the same direction, with 16 ES and two biodiversity parameters considered. However, one challenge reinforced by dealing with numerous ecological parameters is to choose the “good” models, by balancing their complexity, and thus the resources needed to run them, and their informativeness, i.e. the quality and focus of representation of natural processes.

As described in Tables 1 to 4, ecological parameters were modelled individually, leading to the use of a wide range of models: disaggregation of public statistics (e.g. hunting statistics), process-based models (e.g. STREAM for hydrological properties) and analytical models (e.g. RUSLE for erosion losses). We did not use a specific modeling software, as has been done in other ES assessments with for example InVEST, the Integrated Tool to Value Ecosystem Services and their trade-offs (Nelson et al. 2009, Bai et al. 2011), or ARIES, the Artificial Intelligence for Ecosystem Services (Villa et al. 2014, Bagstad et al. 2014), among others. This choice granted us benefits from multiple external collaborations that provided us with specific datasets and expertise on individual models and data sets (at European, national or alpine scales). The use of multiple individual models also increased model adequacy to specificities of the French Alps. As an example, biotic limitation of soil erosion was calculated by adapting the RUSLE (Revised Universal Soil Loss Equation) to mountainous topographic and climatic conditions (Bosco et al. 2009), thanks to the ClimChAlp project (http://www.climchalp.org/), which focused on natural hazard impacts in the context of climate change in the Alpine space. All the same, plant species richness was specifically assessed for the French Alps area, from field inventories and modelled potential ecological niche distributions (Thuiller et al. 2014).

Regarding ecological parameters for which no external collaboration was engaged, we selected models that did not require much specific skills or fine input datasets to be run. For instance, we preferred basing our fodder production estimate on publicly available harvest statistics (AGRESTE http://agreste.agriculture.gouv.fr/), refined by altitude and eco-regions, instead of going through a conversion of orthophotos into i) NDVI (Normalized Difference Vegetation Index), ii) LAI (Leaf area index) and finally iii) biomass estimates. Indeed, even if the second modeling approach could have been followed, gains in yield estimates and in mapping precision did not appear so necessary to the global ES assessment compared to time requirements and to the broad interpretation objectives of this study. Moreover, our large scale of interest basically justified the focus on rougher models.

B. Some geographical issues

When choosing the models and the precision of their outputs, one has to keep in mind the final goal of the study. Global assessments (Naidoo et al. 2008, Costanza et al. 1997) provide valuable information and increase knowledge regarding ecosystem functioning, but decision-making processes at sub-national scale require more complex models and specific inputs (Burkhard et al. 2009). To address our general concern of co-variation between multiple ecological parameters and their links to landscape features in a massif scale perspective, without needing to address local land planning constraints, a patchwork of models differing in their initial scale of focus and in their complexity seemed a good compromise between the number of ecological parameters considered and the resources we could allocate to this assessment.

All datasets were brought to a common 1*1 km resolution, either through the aggregation of finer-scale process information (e.g. protection against gravitational hazards, initially at 25*25m) or by downscaling coarser statistical information (e.g. leisure hunting, by administrative hunting zones). However, as thoroughly explored in England by Anderson et al. (2009), co-variation structures between ES and biodiversity appear sensitive to the spatial resolution of datasets. Their biophysical assessment of three ES and biodiversity concluded that correlations, although presenting similar trends, weakened at finer resolution (4 km^2) compared to coarser ones (100 km^2), while at the opposite overlaps of hotspots increased. As such, our findings could slightly differ if we had decided on alternative common resolution. Nevertheless, we believe that trends would have been conserved as the range of resolutions we dealt with was not as large as the one explored by Anderson et al (2009) and remained comprised between 25 m and 1km. Moreover, we jointly analysed distributions modelled at varying initial extents (e.g. European Union for pollination, and French Alps for plant diversity), thus overlapping outputs of different levels of precision and complexity. Although we used the best models and datasets available, we acknowledged the influence of resolution and initial extent of mapping on spatial associations detected between ecosystem services and biodiversity

An inevitable consequence of our choices is that these results make sense at the scale of the French Alps, meaning that no local extrapolation should be made from them. This argument is supported by the assessment proposed by Anderson et al (2009) (see above), which concluded that relationships between ES and biodiversity were both location specific and sensitive to analysis extent. Indeed, conclusions on the sign and magnitude of associations between ES and biodiversity differed when assessed for Britain as a whole or for smaller windows within the study area. An improvement to our methodology could be to consider “the connectedness of the nested scales” at which ecological parameters occur (e.g. watershed for maintenance of water quality, local landscape for pollination) (Smith et al. 2011). However, this approach would require using hierarchical spatial models that can account for spatial covariance at different resolutions, which was beyond our scope.

C. From reality to mapped variables: what do we actually represent?

A concern additional to ecological parameter selection and to model and scale issues related to what was actually represented by each modelling process. Indeed, the ES concept encompasses both static and dynamic aspects (i.e. stock/status and flows) and can be described according to three distinct facets, further explored in Chapter II: i) potential supply, depending on biophysical capacities of ecosystems to supply an ES, ii) demand, when considering social requirements, and iii) actual supply, expressing the meeting of potential supply and demand with external constraints (as laws, land allocation choices…).

Crossman et al. (2013) called for an explicit ES “accounting definition” that would state its type (e.g. stock, flow, process, function) and its beneficiary (e.g. supply, demand, benefiting/providing area). We described accordingly our set of ecological parameters (Supporting Information S1.A – Parameter characteristics) and concluded on the heterogeneity of those variables. Indeed, we combined stocks (e.g. carbon stocks) with status (e.g. potential for rural tourism) and flows (e.g. hydro-energy potential). Moreover, some ES represented potential supply only (e.g. biological control of pests, plant species richness), potential supply and aspects of demand (e.g. recreation, or wood production, with inclusion of social preferences and constraints), or actual supply (e.g. leisure hunting, protection against rockfalls).

Thus, variables chosen to describe the ecological parameters were able to represent reality according to a certain point of view. For instance, the biophysical ability of ecosystems to supply wood products differs if we assess potential supply (i.e. depending only on biophysical forest characteristics), if we consider demand aspects (i.e. social demand for local timber), or if we expand the analysis to actually harvested volumes by also including other determinants, such as accessibility and economic profitability. The three distributions corresponding to these three descriptions of the same ES would differ, and so would the synergies and trade-offs detected with other ES.

As a consequence, bundles and relationships between ecological parameters need to be interpreted with care. An overlap between a potential ES and an actual one would not convey the idea of an actual overlap but mostly the idea of the suitability of the habitat for supplying both (maybe conditional to specific practises).

Moreover, proxies were used to provide a simplified approach to complex ecological processes (e.g. pollination approximated by habitat suitability for wild pollinators). The use of proxies is known to influence the trade-offs found between ecological parameters (Eigenbrod et al. 2010), but also represents the only option to integrate some ecological parameters. In this study, we kept our proxies as close as possible to the direct variable but could not evaluate the impacts of variable choices on our results.

Overall, the process of selection, modeling and mapping ecological parameters implied multiple choices and led to aggregating non-estimated uncertainties (Smith et al. 2011). Indeed, we were not able to assess uncertainties quantitatively. Reliability of data sources were estimated according to their source (e.g. national agency inventories vs. personal communication) or to the matching between alpine ecosystems and the initial biophysical settings in the case of value transfers. The lack of uncertainty measures remains wide-spread in ES assessment, although we deeply acknowledge the fact that such estimates would be much appreciated for providing a sound basis to their conclusions (Seppelt et al. 2011). Overall, we stress the need for in-depth comprehension of mapped and analysed variables to understand and use the results proposed, as will be discussed in the last section.

IV. Statistical analyses

A. Objectives and methods

Anticipating how environmental changes and management options impact ecological parameters or shape their bundling requires a good understanding of ES and biodiversity interactions. However, "the complex interplay of different ecological processes, dynamic in time and space and often presenting nonlinear behaviours" (Briner et al. 2013) makes this task challenging. Recently, a formalized framework was proposed to guide the quantitative assessment of ES associations (Mouchet et al. 2014, for which I was a co-author – See at the end of the manuscript Appendices from Chapter I (Section A) for the paper). In addition to lexical clarifications, in this paper we proposed the following three-step approach to progress in the exploration of co-variation patterns and determinants: i) detecting ES associations, ii) identifying bundles of ES, and iii) exploring spatial drivers of associations. A main concern of this work was to provide guidance on the adequate analytical tools for answering the questions associated to each of those three steps.

Our French Alps biophysical assessment relied on the three steps from this methodological framework and mobilised various statistical analysis tools (Table 6).

Table 6: Statistical tools mobilised to answer the three-step framework for the quantitative assessment of ecological parameter associations

Step	Scale	Tool	Objective
i) Detecting associations between pairs of EP	Regional	Pearson correlation coefficients	Detect which pairs of ecological parameters are overall positively and negatively associated
		Pairwise overlaps	Add a spatial dimension in the detection of EP pairwise associations
ii) Identifying bundles of EP	Regional	-	Identify bundles of EP by combining their regional pairwise associations
	Sub-regional	Self-organising map	Identify clusters of pixels characterized by similar ecological profiles
iii) Exploring spatial drivers of associations	Landscape	High value clustering Pairwise overlaps	Explore the prominent spatial associations between land cover types and EP
Additional step) Linking landscape heterogeneity and ES diversity	Landscape	Statistics on a 3*3km moving window	Distinguish 4 combinations of high and low landscape heterogeneity and ES diversity
		Chi² tests	Highlight major divergences between combination in distributions of altitude and land cover types

B. On the influence of choices in statistical analyses

A purpose of statistical analyses is to consolidate the assessment of ecological bundles by evaluating the strength or consistency of their associations (Mouchet et al. 2014). Nevertheless, such analyses rely on a set of thresholds and quantitative decisions that often remain poorly discussed and accounted for regarding their implementation. In particular, for this biophysical assessment, we made several assumptions during statistical analyses that influenced their outcomes.

- First, a threshold was required for overlap analyses, as they detected overlapping variables from presence/absence datasets. We chose to transform continuous values into binary ones with a threshold at third quartile, after testing transformation at first quartile and median values. This more selective choice was made to ensure robustness of the results. Nevertheless, external opinions (stakeholder opinions, norms) on the level at which each ES can actually be considered as "well supplied" (i.e. presence value) would have been welcome to increase our analysis adequacy to the alpine context (see for example Gos & Lavorel 2012 or Raudsepp-Hearne et al. 2010).
- Second, a choice was made regarding the number of clusters considered in the self-organising map. We finally assessed the ecological profiles for five clusters, after testing the results of a clustering with 3 to 6 clusters. Here, interpretability of the clusters was favoured by comparing the area they delimited to typical alpine regions (including typical splitting of the massif as Northern and Southern Alps, altitudinal distinctions and broad profiles of human land allocation). Our ability to interpret linked ecological profiles was also conditional to this choice, as distinctions between profiles decreased with the increasing number of clusters.
- Third, we had to determine the size of moving windows used to assess surrounding landscape heterogeneity in land cover types and their richness in ES supplied. The final assessment was performed with a 3*3km window while we also compared results from 5*5km and 11*11km windows. The smallest window was finally preferred upon the others because it logically provided finer and more contrasted results and avoided obtaining a blurry and homogenous pattern over the entire region.
- Fourth, combinations of varying levels of landscape heterogeneity and ES richness required an additional threshold to split distributions between "high" and "low" values. In the absence of external opinion on such threshold, we used the median value as discriminant point to ensure, at least, comparability of the four resulting combinations regarding the area they covered.
- Fifth and last, we could not discuss in the paper presented in Section V all relations obtained. Thus we focussed only on the top 15% values i.e. on those presenting the highest correlation values (in absolute terms), overlap rates and deviation from the null model (for Chi² tests residuals).

Overall, these choices were made to increase the robustness and ease the understanding of statistical analyses. We insist on their influence on results, even if we did not thoroughly quantify it. In particular, thresholds used to distinguish high/low and presence/absence values would gain at being decided after stakeholder consultation. They could be used to account for stakeholder different priorities, thus sticking more closely to the actual benefits people demand from ecosystems (Lamarque et al. 2011, Gos & Lavorel 2012, van der Biest et al. 2014).

V. Results - Assessing bundles of ecosystem services from regional to landscape scale: insights from the French Alps

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References from the paper are available at the end of Chapter I.

Supporting Information is available in the Appendix section at the end of the manuscript (Appendices from Chapter I (Sections C and D)).

Summary

1. Assessments of ecosystem services (ES) and biodiversity (hereafter ecological parameters) provide a comprehensive view of the links between landscapes, ecosystem functioning and human well-being. The investigation of consistent associations between ecological parameters, called bundles, and of their links to landscape composition and structure is essential to inform management and policy, yet is still in its infancy.
2. We mapped over the French Alps an unprecented array of 18 ecological parameters (16 ES and two biodiversity parameters) and explored their co-occurrence patterns underpinning landscape multifunctionality. We followed a three-step analytical framework to i) detect ES and biodiversity associations relevant at regional scale, ii) identify clusters supplying consistent bundles of ES at sub-regional scale, and iii) explore the links between landscape heterogeneity and ecological parameter associations at landscape scale.
3. We used successively correlation coefficients, overlap values and self-organizing maps to characterize ecological bundles specific to given land cover types and geographic areas of varying biophysical characteristics and human uses at nested scales from regional to local.
4. The joint analysis of land cover richness and ES gamma diversity demonstrated that local landscape heterogeneity alone did not imply multifunctionality, while homogeneous landscape could be multifunctional.
5. **Synthesis and applications**: Bundles of ES and biodiversity parameters are shaped by the joint effects of biophysical characteristics and of human history. Due to spatial congruence and to underlying functional interdependencies, ecological parameters should be managed as bundles even when management targets specific objectives. Moreover depending on the abiotic context multifunctionality can arise either from deliberate management in homogeneous landscapes or from spatial heterogeneity.

Keywords

Biodiversity
Biophysical assessment
Ecosystem service association
Synergy and trade-off
Landscape heterogeneity
Natural resources policy
Multi-scale assessment

1 Introduction

The links between landscapes, ecosystem functioning and human well-being, as captured by the ecosystem service concept, have emerged as a powerful bridge between science and policy (Perrings *et al.* 2011). Relationships between ecosystem services (hereafter ES), as well as between ES and biodiversity, can be understood by identifying which co-vary positively or negatively. Evaluating their repeated associations goes beyond the assessment of a static snapshot and enable concluding on "synergies", that can be actively stimulated, and "trade-offs", that should be anticipated and limited, respectively (Raudsepp-Hearne, Peterson & Bennett 2010, Mouchet *et al.* 2014; Verkerk *et al.* 2014). In particular, the consistent associations in time and/or space between multiple services, known as "bundles" of ES (Raudsepp-Hearne, Peterson & Bennett 2010), differentiate areas supplying the same magnitude and types of ES as a result of a shared socio-ecological profile. Considering ES bundles in natural resources management is thus ecologically relevant and should facilitate the communication of the complexity of ecological interactions to stakeholders (Van der Biest *et al.* 2014).

ES assessments increasingly use the concept of landscape multifunctionality, understood as "the capacity of a landscape to simultaneously support multiple benefits to society from its interacting ecosystems", relying on the "joint supply of multiple ES at the landscape level" (Mastrangelo *et al.* 2014). Landscape heterogeneity closely links to multifunctionality (Brandt 2003) and appears easy-to-access for scientists and easy-to-grasp for stakeholders (Laterra, Orúe & Booman 2012). Yet, the extent and generality of spatial or functional associations between landscape heterogeneity and multifunctionality are still debated (Anderson *et al.* 2009; Mastrangelo *et al.* 2014). In this context, a better understanding of associations among ES and of their relations to spatial patterns of underlying biophysical variables is needed for more effective land allocation and management (Briner *et al.* 2013).

To progress in this endeavour, Mastrangelo et al. (2014) proposed two alternative perspectives on landscape multifunctionality. First, spatial approaches can detect pattern-based multifunctionality. Often focusing on land cover, they identify bundles from spatial coincidence and can guide spatial planning and priority setting. However, no fine understanding of ecological processes and interactions is gained. Second, functional and spatio-functional approaches can detect process-based multifunctionality. Both approaches explicit model drivers of individual ES, the latter being additionally spatially explicit. They increase the ecological understanding of relationships between ES and can support optimal management solutions balancing their supply levels. The availability of ecological data and models guides the choice between these three approaches. Other approaches exist but require stakeholder involvement, which was beyond the scope of this study.

In this study, we applied in the French Alps a spatial approach for a pattern-based multifunctionality assessment at regional scale. Of the several ES assessments in mountain regions (reviewed by Grêt-Regamey, Brunner & Kienast 2012), several have highlighted the role of spatial heterogeneity resulting from natural and human factors (Briner *et al.* 2013) for supporting high multifunctionality (Grêt-Regamey, Brunner & Kienast 2012). The European Alps encompass a high diversity of ecosystems, species and landscapes, due to broad and often steep gradients of topography, soils, altitude and climate (Tappeiner, Borsdorf and Tasser 2008). Within their range, a long history of human-nature interrelations has shaped cultural landscapes (EEA 2010), and so influenced ecological functioning. This directly affects the many ES supplied to their population and to many living beyond them (EEA 2010). Yet, in-depth joint biophysical assessments of ES and biodiversity are still scarce (Grêt-Regamey, Brunner & Kienast 2012).

To address this need, we explored the following hypotheses: 1) different bundles of ecological parameters can be identified and linked both to diverse biophysical conditions and to land allocation and management choices, and 2) heterogeneous landscapes provide richer sets of ES than homogeneous ones. For this, we mapped an unprecented array of 16 ES and two biodiversity parameters (regrouped as ecological parameters henceforth) using ecological models. We then analysed their joint variations as an expression of multifunctionality, and lastly explored and characterized their spatial patterns at various scales from the entire region to the landscape.

Figure 1 summarises our research questions and analytical framework following the three-step framework by Mouchet *et al.* (2014) to: i) detect ES and biodiversity associations relevant at regional scale, ii) identify clusters supplying similar bundles at sub-regional scale, and iii) explore the links between landscape heterogeneity and ecological parameter associations at landscape scale. This third step analysed both how ecological bundles overlap with dominant land cover types, and how ES diversity relates with landscape heterogeneity. We explicitly related all analyses to potential application by discussing their scale-specific relevance to stakeholders concerned by natural assets in the French Alps.

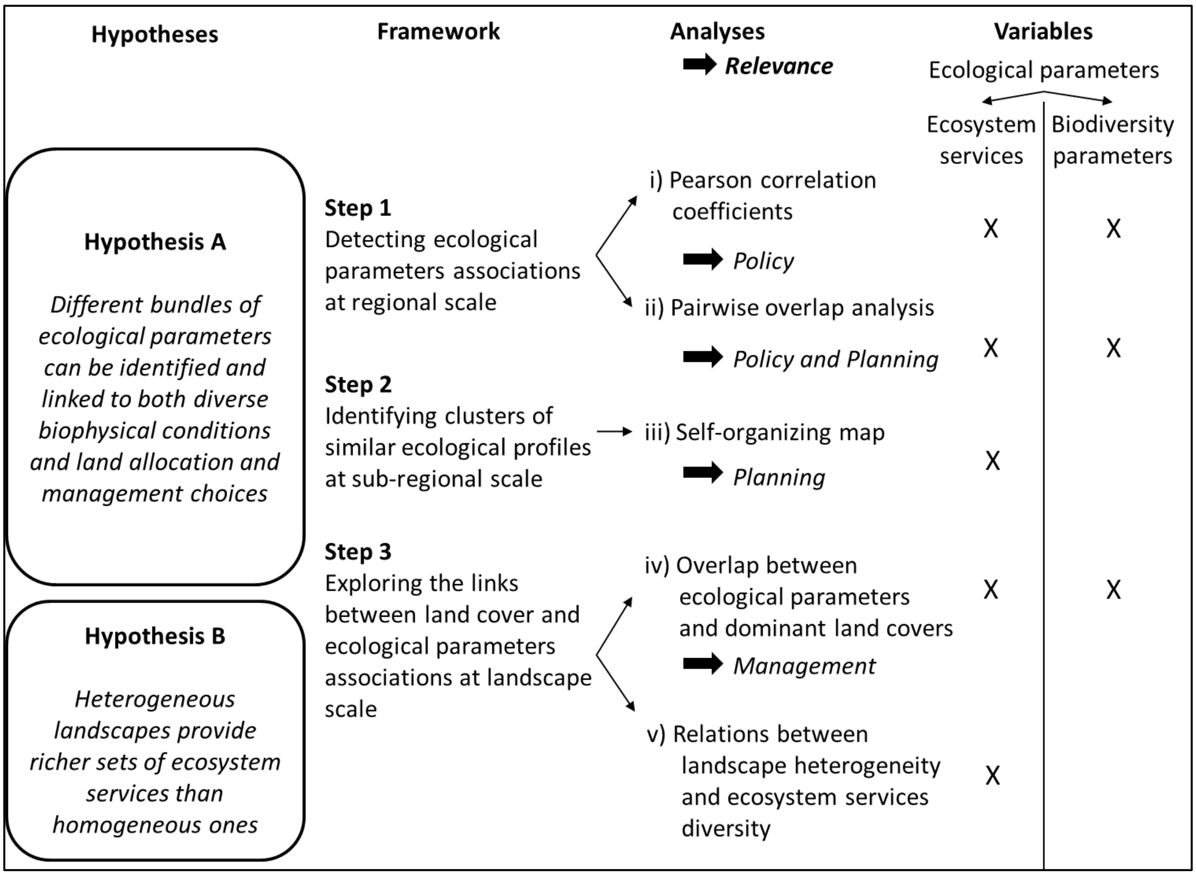


Figure 1: Analytical framework and hypotheses tested.

2 Materials and methods

2.1 Study region

Our analysis focused on the French Alps as defined by the Alpine Convention (SPCA 1991) covering 52 149 km² over the western part of the Alpine arc. The complex topography formed by Tertiary tectonic activity followed by glaciations encompasses elevations from below 100 m to 4810 m (Mont Blanc). Latitudinal climate and vegetation gradients have had historical consequences on social dynamics and economic activities, resulting in the common separation into the Northern and the Southern Alps. A secondary longitudinal climatic and geological gradient runs from the western Atlantic influence, known as the Prealps, to continental climate in the inner Alps. This geographic diversity is responsible for the high variety of biodiversity, ecosystems and ES across the entire area compared to European averages (Tappeiner, Borsdorf & Tasser 2008).

Based on Corine Land Cover 2006 Level 1 categories (EEA 2012), the French Alps are dominated by forests and semi-natural areas (67% of the region). Arable lands are mainly concentrated in the western broad valleys and piedmonts (27% of the region), while artificial areas cover only 5% of the region. This leads to a clear distinction between high-density urban areas surrounded by intensive agriculture in the valleys and more isolated or higher rural areas (Tappeiner, Borsdorf & Tasser 2008).

2.2 Modelling and mapping ecological parameters

- **Selection of ecological parameters: ES and biodiversity**

Following consultation with scientists and local collaborators, we selected four provisioning, five cultural and seven regulating ES, and two biodiversity parameters (plant and vertebrate diversity), encompassing most services relevant to the region from ecological, social and economic points of view (Table 7).

Table 7: ES and biodiversity parameters considered in the assessment of ecological relationships over the French Alps. Abbreviated names between brackets are those used for all analyses. Type specifies: P = provisioning service, C = cultural service, R = regulating service, B = biodiversity parameter.

Type	Parameter	Description (unit)	Sources
P	Agricultural production (**crop**)	Yields for annual crops, vineyards and orchards (kg/ha/yr)	Agreste 2009
P	Forage production (**fodd**)	Yields of pastures, meadows and mountain grasslands (kg dry matter/ha/yr)	Agreste 2009 - Supporting Information S1.B
P	Wood production (**wood**)	Potential woody biomass supply for stemwood and logging residues (Gg dry matter/km²/yr)	Verkerk *et al*. 2011; Brus *et al*. 2012; Elbersen *et al*. 2012
P	Hydro-energy potential (**hydro**)	Theoretical potential hydroelectric power delivered by river basin (classes)	Agence de l'eau RMC 2008
C	Recreation potential (**recre**)	Recreation potential for daily recreation (index)	Paracchini *et al.* 2014
C	Tourism (**tour**)	Territorial capital of rural tourism involving overnight stays (index)	Paracchini & Capitani 2011; Maes *et al.* 2012, Paracchini *et al.* 2014
C	Leisure hunting (**hunt**)	Density of shot wild ungulates (number of animals/km²/yr)	Convention with « Réseau Ongulés Sauvages ONCFS / FNC / FDC » Supporting Information S1.C
C	Protected plant species (**protp**)	Species richness for 45 protected plant species with Red List status critical, endangered and vulnerable (number of species/km²)	Thuiller *et al.* 2014
C	Protected vertebrate species (**protv**)	Species richness for 107 protected vertebrate species with Red List status critical, endangered and vulnerable (number of species/km²)	Maiorano et al. 2013
R	Erosion mitigation (**eros**)	Biotic contribution to erosion risk mitigation (classes)	Bosco *et al.* 2008; Bosco *et al.* 2009
R	Protection against rockfalls (**rock**)	Ability of forests to decrease rockfall hazard and protect sensitive human areas (index)	Berger *et al.* 2013
R	Chemical water quality regulation (**wql**)	Nitrogen retention capacity by river basin (tN/km/year)	Grizzetti & Bouraoui 2006
R	Physical water quantity regulation (**wqt**)	Relative water retention enabling flood regulation (index)	Stürck, Poortinga & Verburg 2014
R	Biological control of pests (**cbiol**)	Species richness for 110 vertebrate species providing natural pest control (number of species/km²)	Civantos et al. 2012; Maiorano et al. 2013
R	Pollination (**poll**)	Relative landscape suitability for pollinators (index)	Zulian, Maes & Paracchini 2013
R	Carbon storage (**csto**)	Sum of carbon stocks from above-ground and below-ground biomass, dead organic matter and soils (tC/km²)	Martin et al. 2011; Meersmans et al. 2012a, 2012b; Supporting Information S1.D
B	Plant diversity (**plant**)	Species richness for 2748 plant species using their potential ecological niche distributions (number of species/km²)	Thuiller *et al.* 2014
B	Vertebrate diversity (**vert**)	Species richness for 380 vertebrate species using their potential ecological niche distributions (number of species/km²)	Maiorano *et al.* 2013

- **Modelling ecological parameters**

Depending on model and data availability, the 18 ecological parameters were modelled using methods ranging from disaggregation of public statistics (e.g. hunting statistics) to process-based models (e.g. STREAM for hydrological properties - Stürck, Poortinga & Verburg 2014) and analytical models (e.g. RUSLE for erosion losses - Bosco *et al.* 2009) (Table 7). To allow joint analysis, all ecological parameters were rescaled to a 1*1km resolution, through aggregation of finer-scale process information (e.g. protection against gravitational hazards) or downscaling of coarser statistical information (e.g. leisure hunting). Supporting Information S1.A provides standardised descriptions for all ecological parameters (Crossman *et al.* 2013), with additional information on methods and data sources following Martínez-Harms & Balvanera (2011).

Our selection comprised both potential values for ecosystem parameters, based on the natural capacity of ecosystems, and actual values, considering the actual benefits to society (Van der Biest *et al.* 2014). Then, the observed association between parameters does not necessarily imply that they are actually supplied jointly, but merely that the ecosystem has the potential for supplying both. For instance, an association between potential plant habitat and actual crop production would not mean that croplands host a high biodiversity, but only that natural conditions suitable for cropping are also conducive to plant diversity, whether agricultural practices support their actual coexistence or not. Additionally, three types of parameters were combined depending on their nature and data availability: stock (e.g. number of species/km²), flow (e.g. tons of wood harvested/year) or status (e.g. relative capacity to buffer floods).

Land cover categories used to analyse the joint occurrence of ecological parameters were those of Corine Land Cover 2006 (CLC 2006) aggregated at 1km*1km to match the resolution of ES data. For altitude we used the 50m French digital elevation model BD-ALTI® IGN.

2.3 Statistical analyses

Spatial data processing was done using ArcGIS 10.0 and statistical calculations were carried out using the statistical software R 2.15.

After an initial standardization and normalisation phase, data analyses followed three successive steps aiming to: i) detect consistent associations between ecological parameters at regional scale, ii) identify clusters at sub-regional scale and describe their spatial patterns and geographical determinants, and iii) explore the links between landscape and ecological parameter local associations. Two points need attention for the interpretation of results. First, we insist that the bundles we detected rely on spatial coincidence rather than on identification of common functional drivers. Second, as we considered jointly potential and actual ES parameters, associations do not necessarily reflect synergies and can even relate to conflicts as further discussed below.

2.3.1 Data transformation

As ecological parameters had different units and scales (Table 7), we made the range and the variability of values comparable across variables by re-scaling each dataset to a common, unit-less [0-1] interval by subtracting from each value the minimum value observed for the dataset and then dividing by the difference between the observed maximum and minimum values (Paracchini *et al.* 2011).

Although normality of the datasets was not required since we did not perform any parametric test, we limited skewed variances that could respond heterogeneously to statistical analyses by logarithm or square-root transformation after visual examination of the frequency distribution.

Finally, binary presence and absence datasets were obtained with a threshold at third quartile after removing zero values, chosen following a comparison with thresholds at first quartile and median (results not shown).

In the presentation of results for the following analyses, we comment only the 15% largest values to focus on prominent features, resulting in specific thresholds for Pearson coefficients, overlap ratio and Chi² test residuals.

- **Step 1: Detecting consistent associations at regional scale**

Two complementary analyses were used to detect consistent associations between ecological parameters at regional scale (Egoh *et al.* 2009).

First, we used Pearson's coefficients to test positive and negative associations between pairs of ecological parameters at the scale of the entire study area.

Second, spatially consistent associations between pairs of ecological parameters considered as binary presence / absence were detected using an overlap index (Gos & Lavorel 2012). For pixels with "present" ecological parameters, we calculated the fraction *O* of pixels in the smaller dataset that overlapped with the second one. *O* can vary from 0 (no overlap) to 1 (all cells of the smallest dataset overlapping with the second one).

- **Step 2: Identifying clusters at sub-regional scale**

In order to explore sub-regional ES associations (Anderson et al. 2009), we used Kohonen's algorithms to build a Self-Organising Map (SOM) delineating five clusters of pixels with specific ecological profiles, each supplying a consistent bundle of ES. The number of clusters represented the best compromise between analysis complexity and interpretability. We analysed their geographic distributions, altitude and land cover patterns.

- **Step 3: Exploring links with land cover at landscape scale**

Links between ecological parameters and landscape were investigated by: i) the overlaps between individual ecological parameters and dominant land cover types, and ii) the relation between ES diversity and landscape heterogeneity.

High value clusters for individual ecological parameters and land cover types were detected with ArcGIS Hot Spot Analysis tool parameterized to calculate Getis-Ord Gi* statistics using the "Distance Band or Threshold Distance" cut-off to a window of 3*3km. Significant p-values were returned when observed spatial clustering was greater than expected for a random distribution, avoiding the selection of isolated pixels of high values or outliers. Each variable was then transformed into a binary dataset, attributing a value of 1 for clusters with z-scores significant at 10% minimum and 0 otherwise. Pairwise overlap analysis detected spatial matches between clusters of high value for ecological parameters and for land cover types.

Local landscape heterogeneity and ES diversity were assessed by affecting to the central pixel of a moving 3*3km window the number of unique land cover types (ArcGIS Focal Statistics tool with the "Variety" option) and the number of distinct ES (equivalent to a gamma index). In absence of socially relevant thresholds, the distributions of these two variables were split between high and low values according to the median, leading to four possible combinations of low/high landscape heterogeneity and gamma index. Chi² tests were used to detect major divergences between actual distributions of altitude and land cover type in the different combinations, compared with their frequencies over the whole French Alps taken as null model (Chi² tests significant at 5%, deviation of residuals greater than 10). Pairwise overlaps between pixels from the four categories and distributions of specific ES were also tested.

3 Results

3.1 Associations at regional scale

Results from Pearson coefficients (Supporting Information S2.A) and pairwise overlap analysis (Supporting Information S2.B) were highly consistent, showing some strong positive associations among ecological parameters and with specific land cover types (Supporting Information S2.D). Based on these we identified three bundles (Figure 2). Bundle A encompassed multiple positive associations among three ES overlapping with agricultural areas: crop production, plant diversity and maintenance of water quality, the latter being also associated with hydro-energy production. Bundle A was negatively correlated to cultural ES (plant diversity vs. recreation and tourism, and crop production vs. recreation). Bundle B encompassed multiple positive associations among three ES overlapping with forests: wood production, carbon storage and regulation of water quantities. Wood production and carbon storage were also correlated with vertebrate diversity, while carbon storage was additionally correlated with erosion mitigation. Bundle B also overlapped with protection against rockfalls and recreation. The negative correlation between carbon storage and plant diversity resulted in a negative association between bundles A and B. Bundle C encompassed multiple positive associations among biological control, protected vertebrate diversity and vertebrate diversity, the latter also presenting a positive correlation to bundle B (with wood and carbon storage). Bundle C also incorporated erosion mitigation through its overlap with biological control. Lastly, protected plant diversity, which positively overlapped with bundle A through plant diversity, correlated negatively with both bundles B (through wood production and carbon storage) and C (through vertebrate diversity and biological control).

Regarding land cover, although some groups of ecological parameters were tightly associated with specific land cover types (bundles A and B with agricultural areas and forests respectively), others from the same bundles overlapped with distinct types: in bundle A hydro-energy production and plant diversity overlapped with grasslands and open spaces, and artificial areas respectively; in bundle B protection against rockfalls and recreation overlapped with open spaces, with recreation also overlapping with grasslands. Conversely individual ecosystem parameters could overlap with multiple land cover types as for biological control (bundle C) with agricultural areas, wetlands and semi-natural open areas (also overlapping with pollination).

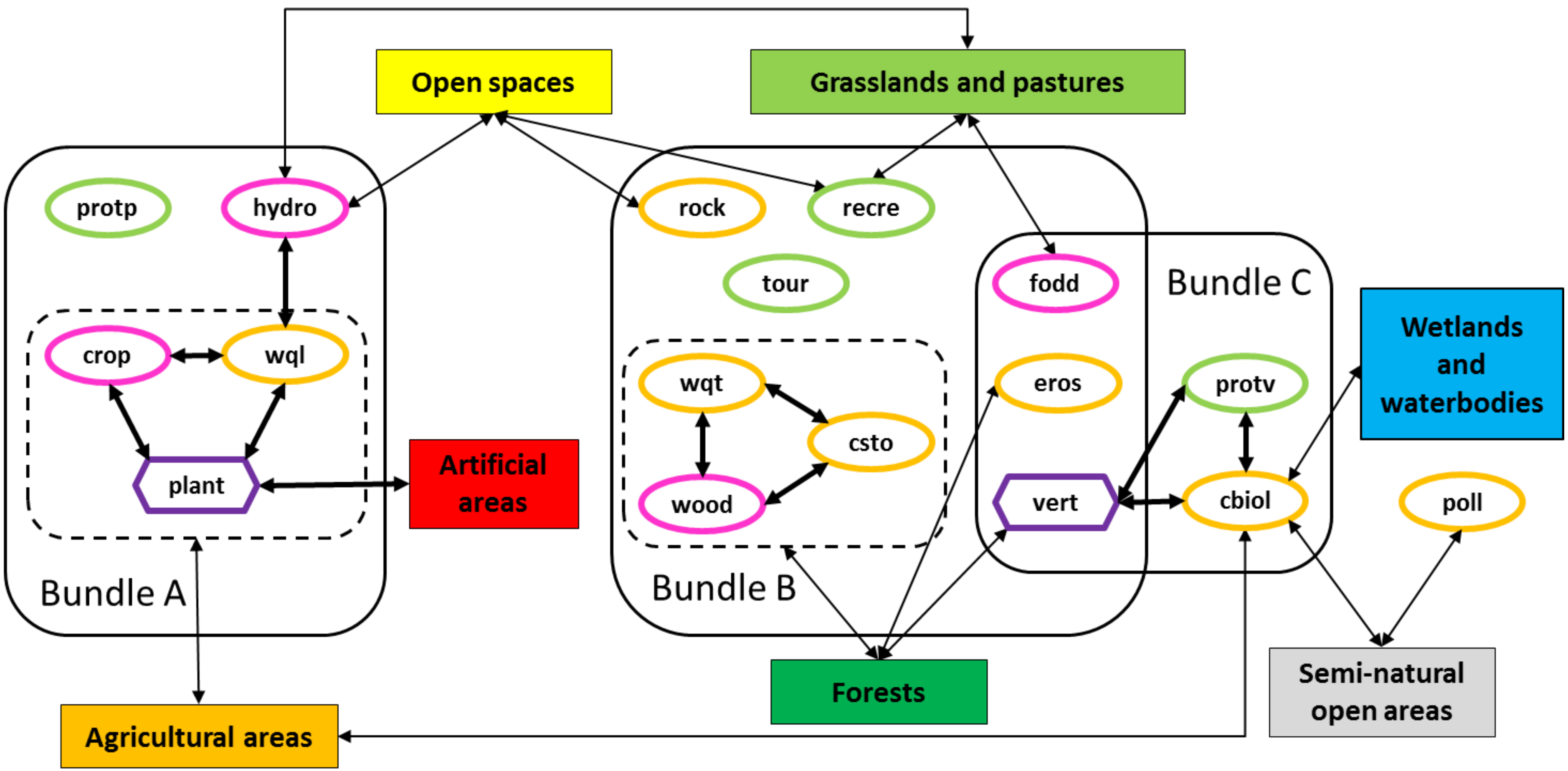


Figure 2: Bundles of ecological parameters (ES and biodiversity parameters) and overlaps with dominant land covers. Bundles were identified by Pearson coefficients and pairwise overlaps (solid lines). Bold arrows: consistent associations between parameters for both analyses. Associations with land cover types were identified through overlaps between ecological parameters and land cover high value clusters (plain arrows to individual parameters or to multiple parameters encompassed in dotted lines). Biodiversity parameters are presented as hexagons (purple border) and ES as ellipses (pink border: provisioning services, green border: cultural services; orange border: regulating services). See Table 7 for abbreviations.

3.2 Clusters at sub-regional scale

Five clusters of ES were identified by the self-organizing mapping algorithm (Fig. 3 - see Supporting Information S2.C for altitudinal and land cover distributions).

Cluster 1 (red pixels) contributed strongly to crop production, biological control, protected vertebrate species richness and maintenance of water quality. Mainly located at low altitudes in piedmonts and in the main valleys, it covered the highest proportions of urban and agricultural lands, associated to gentle climate and topography.

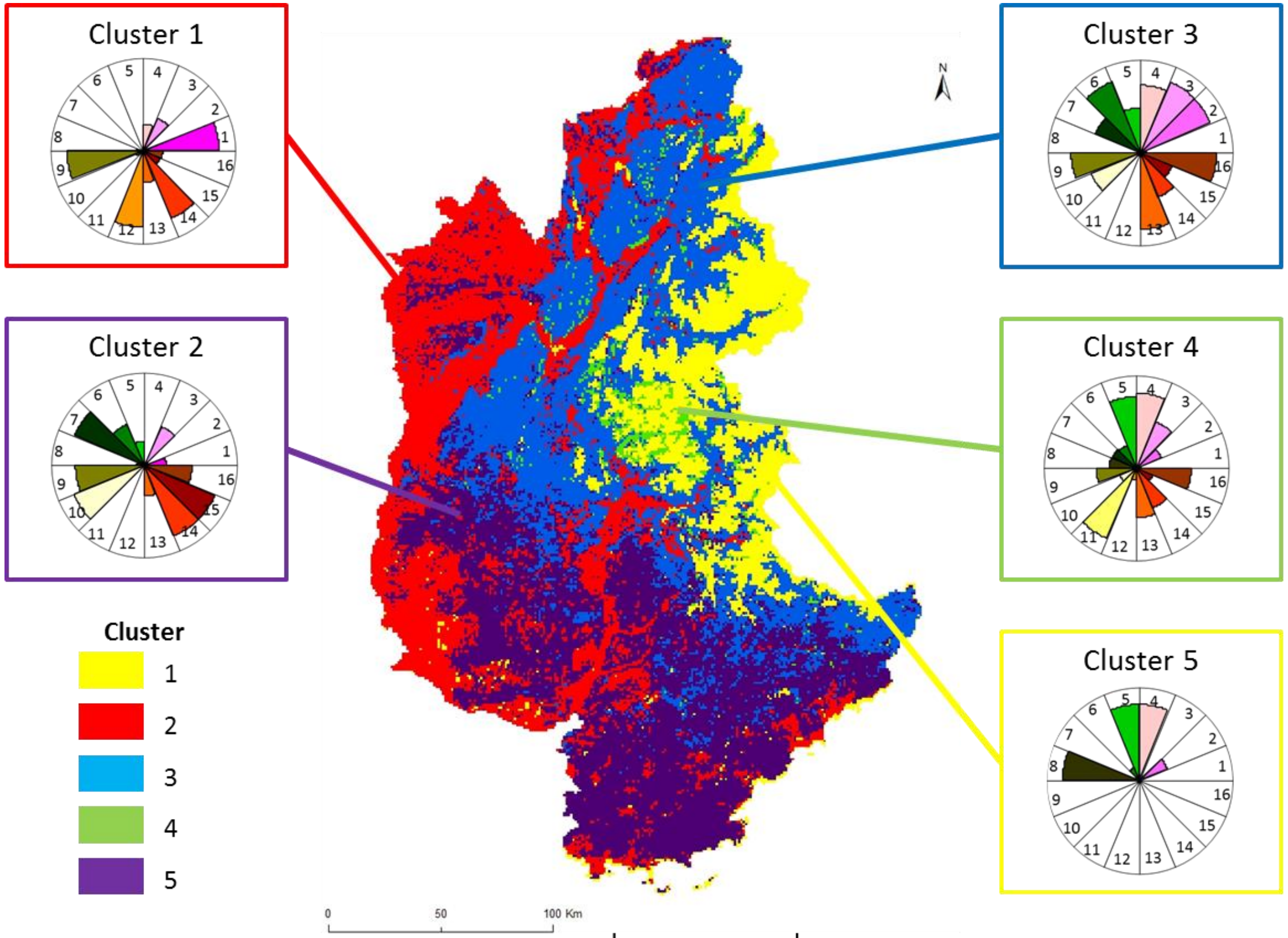
Clusters 2, 3 and 4 presented richer bundles of ES and encompassed landscapes of intermediate altitude with more than 50% forests.

Cluster 2 (purple pixels) concentrated in the Southern Alps, contained few grasslands but a high proportion of semi-natural and open areas. It supplied mostly cultural and regulating services, with strong levels of fauna-related services (leisure hunting, protected vertebrate species, biological control of pests and pollination) reflecting the suitability of such (semi-)natural ecosystems as habitats and resources for wildlife. Biotic contribution to erosion mitigation was also high due to high environmental exposure.

Cluster 3 (blue pixels) contained the highest proportion of grasslands and pastures, which along with forests supplied high levels of provisioning services (forage production, wood production and hydro-energy potential). Cultural services (recreation, tourism, leisure hunting and vertebrate protected species) and forest-related regulating ES (water quantity regulation and carbon storage) were also well supplied. Although less prominent than in cluster 2, biotic contribution to erosion mitigation, biological control of pests and pollination were also characteristic regulation services.

Cluster 4 (green pixels), restricted to a small area of the Central Alps, combined forests with open areas with scant vegetation cover. Its particularly high level of protection against rockfalls by forests was explained by its location at the interface between high altitude, steep cluster 5 areas uphill of cluster 3 areas containing valued and managed spaces.

Cluster 5 (yellow pixels) supplied a restricted set of ES, mainly hydro-energy potential, recreation potential and protected plant species. Its high altitude location in the eastern part of the French Alps, covered mainly by open spaces with little or no vegetation, suggested that overall harsh climatic conditions, not favourable to vegetation development, led to a low biotic contribution to ecological processes and limited ES supply.



Provisioning	1 – crop	2 – fodd	3 – wood	4 - hydro	-
Cultural	5 – recre	6 – tour	7 – hunt	8 - protp	9 – protv
Regulating	10 - eros	11 – rock	12 - wql	13 – wqt	14 - cbiol
	15 – poll	16 - csto	-	-	-

Figure 3: Self-organizing map with five clusters and related ecological profiles (values standardised to 0-1). See Table 7 for abbreviations.

3.3 Landscape combinations of land cover heterogeneity and ES diversity

The four combinations of landscape heterogeneity and ES gamma index (Fig. 4) showed that high landscape heterogeneity did not necessarily beget high ES richness (see Supporting Information for Chi² tests residuals: S2.E for land cover distributions, S2.F for altitude distributions, and S2.G for overlap with ES).

Low values for landscape heterogeneity and gamma index (combination LL, grey pixels) covered 22% of the French Alps, either in agricultural areas at low altitude (0-500m) or in open spaces at high altitude (>2000m). Conversely, homogenous landscapes with a high gamma index of ES (combination LH, yellow pixels, 18% of the region) were over-represented in forests at intermediate altitudes (1000-1500 m), regardless of forest type (broad-leaved, coniferous and mixed forests) (data not shown).

Artificial areas and semi-natural areas were over-represented and forests under-represented in heterogeneous landscapes supplying few ES (combination HL, blue pixels, 19% of the region). Conversely, grasslands and pastures and semi-natural areas were over-represented but open spaces under-represented in heterogeneous multifunctional landscapes (combination HH, red pixels, 41% of the region). Among heterogeneous landscapes open spaces and artificial areas were over-represented and forests under-represented in areas of low (HL) compared to high ES supply (HH).

Lastly, the two combinations with diverse ES (LH and HH) differed in the strength of their overlaps with ecological parameters. While homogenous multifunctional forest landscapes (LH) presented the highest overlaps with parameters from bundle B (carbon storage, wood production, recreation and regulation of water quantities), heterogeneous multifunctional landscapes (HH) had strong associations with ecological parameters from all bundles, except for crop production, protected plant species and plant diversity from bundle A.

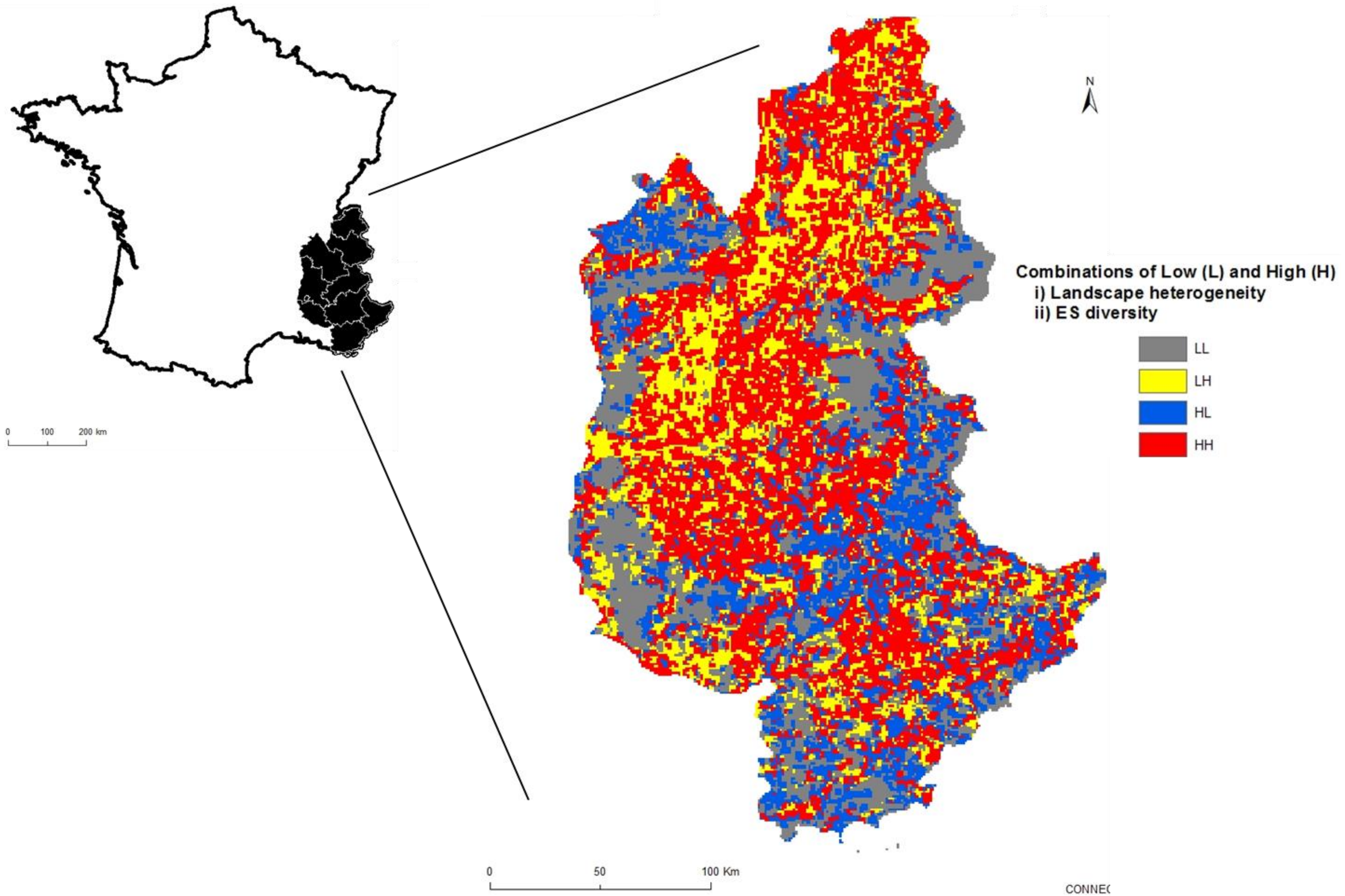


Figure 4: French Alps - Combined landscape heterogeneity and ES gamma index. LL: low landscape heterogeneity and low gamma index, LH: low landscape heterogeneity and high gamma index, HL: high landscape heterogeneity and low gamma index, and HH: high landscape heterogeneity and high gamma index.

4 Discussion

Our multi-step analysis showed how multifunctionality can be explored by detecting consistent associations between ecological parameters at nested scales, from regional bundles to sub-regional clusters and the investigation of their links to local landscape heterogeneity. In the following, we highlight how our results could be appropriated by managers and policy makers in the French Alps (Fig. 1).

4.1 Policy-relevant correlations between ecological parameters at regional scale

Three main factors drove associations between ecological parameters. First, positive correlations between forest-related ES confirmed the multifunctional role of forests, widely promoted in policy (European Commission 2013). Second, strong relationships between biological control and protected vertebrate species were explained by a set of 19 common service-providing species. Third, positive correlations between diversity of vertebrate or plant species and several ES (e.g. wood production or crop production, respectively) relate to specific land covers (e.g. forests or agricultural lands) that simultaneously supply habitats for species and ES. Such associations should be carefully interpreted because these are only potentially suitable habitats. Anderson *et al.* 2009 argued that "this spatial coincidence [between crop production and biodiversity] is likely to be to the detriment of biodiversity", as confirmed by widespread conflicts between production and biodiversity conservation (Maskell *et al.* 2013 for agriculture; Verkerk, Zanchi & Lindner 2014 for forestry). Furthermore, policy promoting cultural services like nature tourism in the French Alps may not warrant biodiversity protection either, as, consistent with England (Anderson *et al.* 2009; Maskell *et al.* 2013), cultural services were negatively correlated to plant diversity. With these regional-scale correlation analyses, we recommend to consider all bundle parameters, and in particular biodiversity, even in policies targeting restricted objectives. In the French Alps, such knowledge could reinforce policy orientations of the Alpine Convention (SPCA 1991) or the Northern Alps planning directive. Nevertheless, despite their interest, correlation analyses cannot warrant causal relationships, requiring careful expert interpretation.

4.2 Spatial associations of ecological parameters and bundles for planning

Incorporating a spatial dimension to ES assessments is a major asset to detect regional specificities and support land planning (Crossman *et al.* 2013).

First, some of the bundles detected by ES overlaps are already incorporated into planning. Alpine forestry guides (e.g. Gauquelin & Courbaud 2006) and forestry regional strategic plans recommend carbon storage, protection against rock falls and mitigation of water flows as joint objectives. Likewise, the overlap between crop production and regulation of water quality is well-known (e.g. Laterra, Orúe & Booman 2012; Qiu & Turner 2013) and is integrated by regional planning for sustainable farming in France and in Britain for example. While this trade-off raises less concerns for the Alps than in more intensive agricultural regions, the sensitivity of mountain ecosystems to human perturbations (EEA 2010) and their role as water towers for surrounding regions (Grêt-Regamey, Brunner & Kienast 2012) are two critical reasons for attention. Second, our analyses revealed overlaps which to our knowledge are less considered in planning. For instance, the overlap between fodder production and regulation of water quantity is seldom targeted by specific measures in the French Alps, despite the known benefit of maintaining grasslands for regulation of water flows. Thus, as for biodiversity, non-provisioning services must be considered explicitly in natural resources planning for long-term sustainability (Maskell *et al.* 2013), as their supply is interlinked with those from the same bundle.

Self-Organizing Mapping complemented overlap analyses by characterising five sub-regional ecological clusters. These clusters were visually linked to commonly described eco-regions of the French Alps. In addition to these biophysical patterns, historical land uses should also be considered to better understand these clusters (Tappeiner, Borsdorf & Tasser 2008). For example, the Southern Alps have undergone a significant decline in their rural population since World War II, leading to agricultural area abandonment and explaining the shift from crop and pasture production to forest-based ES (Cluster 2).
Such description and mapping of ES clusters at sub-regional scale has strong potential for increased appropriation of ecological relations by stakeholders involved in planning, conditional to in-depth analysis for each sub-region before actual decision making. Also, administrative boundaries can be useful mapping units coherent with social management and decisional units to be added in the clustering process (Raudsepp-Hearne, Peterson & Bennett 2010). We suggest applying sequentially unconstrained and administratively-constrained approaches to first account for internal ecological diversity that is not congruent with administrative boundaries, and then incorporate the operational scale for land planning (e.g. municipalities).

4.3 Considering landscape-scale linkages between land cover and ecological parameters for management

High values of specific ecological parameters were linked to either a specific land cover (e.g. carbon storage to forests), or to multiple land covers (e.g. biological control of pests to wetlands, agricultural areas and semi-natural open areas). Therefore, the supply of multiple services would require “an area large enough to encompass the spatial heterogeneity in service supply” (Qiu & Turner 2013). However, high value clusters attributed to a dominant land cover may contain a diversity of land covers, as for the overlap found between artificial areas and plant diversity, which reflected favourable wetland and agricultural fragments within areas dominated by artificial land cover.
Overlaps between land covers and ES provide the basis for region-specific look-up matrices proposed to support landscape analysis and management (Burkhard, Kroll & Müller 2009). Consistent with an expert-based assessment in a German peri-urban area (Burkhard, Kroll & Müller 2009), we found a high combined capacity of forests for erosion regulation, carbon storage and wood production. However our results diverged for agricultural areas which, probably due to less intensive management in the Alps, had high rather than low water quality regulation.
Overlap analysis could support locally-tailored management schemes. Current recommendations in the Alps already incorporate some of the relations we found. For instance, the overlap of both fodder production and recreation potential with grasslands and pastures justified the subsidies by municipalities to livestock grazing and mowing to maintain open landscapes with extensive agriculture that provide naturalness and recreational attractiveness (see Schirpke, Tasser & Tappeiner 2013 for Austria). Other associations not yet included in management strategies would gain in being made explicit to local decision-makers. For instance, we confirmed the relevance of productive forests and grasslands for hydro-energy production but, to our knowledge, vegetation cover is not yet incorporated into watershed management in the French Alps, partly due to a lack of available robust evidence for impacts.

Lastly, the understanding of bundles of ES needs to be supported by overlap analyses with land cover in addition to overlaps among ecosystem properties, as land cover is the first entry to planning and management.

4.4 Relationships between multifunctionality and landscape heterogeneity

Overall, we did not find a unidirectional relationship between landscape compositional heterogeneity and ES richness for the French Alps, which highlights three issues for management.

First, we explain the low ES richness of homogeneous landscapes (LL) by two mechanisms: i) specialization of ES due to management in lowland agricultural areas (Laterra, Orúe & Booman 2012), and ii) biotic limitation and specialization of ES in high altitude open ecosystems.

Second, forest landscapes, although spatially homogenous, supplied a high diversity of ES (LH), though necessarily more restricted than that of highly multifunctional heterogeneous landscapes (HH). We suggest that this multifunctionality reflects both ecological adaptation to current environmental conditions and historical management combining diverse objectives (Courbaud *et al.* 2010).

Third, mosaic landscapes were either linked to low or high multifunctionality. These alternative patterns may be explained by the contrast between artificial areas and open spaces, over-represented in the former case (HL) and unfavourable to the supply of multiple ES, and forests and grasslands, over-represented in the latter case (HH) and favourable to multifunctionality.

Our results demonstrated that homogeneous landscapes can be multifunctional under specific conditions. Such findings could feed debates on landscape design (Maskell *et al.* 2013). However we considered land cover categories as homogeneous across the French Alps, ignoring significant variations due to management and biophysical gradients (e.g. variations in tree species and age-structure in forests). Agri-environment schemes explicitly managing landscape heterogeneity are required to increase (or even create) benefits for farmland biodiversity (Mitchell, Bennett & Gonzalez 2014). In line with this argument, we call for a broader inclusion of landscape patterns for agricultural, forestry, touristic and urban planning.

4.5 Conclusion

Our study explored pattern-based multifunctionality reflecting the repeated coincidence between ecological parameters and landscape features. Its main strength is to promote the management of ES and biodiversity as bundles rather than as individual targets. Bundles arose from the joint effects of two factors. First, biophysical characteristics defined the constraints (e.g. temperature or slope limitations restricting bundles at high altitudes) and opportunities (e.g. favourable abiotic conditions for wild species and for ecological functioning in the Southern Alps) for potential joint supply. Second, bundles have been shaped through human history by land allocation and management choices. The resulting bundles and their relations to landscape features may be generalizable to biophysically and socially comparable regions.

Our analysis supports the explicit consideration of bundles in management, and in particular the integration of biodiversity and regulating services even in policies targeting other objectives. Current management already considers such bundles, such as the joint supply by alpine forests of carbon storage, protection against rock falls and mitigation of water flows. Others such as the association between forage production and regulation of water quantities in extensive grasslands would deserve consideration. Additionally multifunctionality can depending on the abiotic context arise either from deliberate management in homogeneous landscapes or from spatial heterogeneity. Such solutions will require ecosystem-based management at landscape scale, and may be generalizable.

We stress the interest of complementing our results by identifying functional mechanisms underlying associations, which would foster a process-based approach of multifunctionality (Mastrangelo et al. 2014). However increased availability of models (e.g. phenomenological or trait-based models) and data at fine resolution over regional geographical extents (species distributions – abiotic properties) precondition such progress.

VI. Synthesis

This chapter was dedicated to the biophysical assessment of 18 ecological parameters over the French Alps. We explored sequentially four questions (Section I) and could conclude on our ability to:

1) Map the individual distribution of each ecological parameter as biophysical values,
2) Detect associations between pairs of ecological parameters, identify how they bundled at regional scale and further characterise the ecological profiles of five clusters at sub-regional scale,
3) Relate local landscape features (altitude, land cover types) to ecological parameters and to their associations,
4) Describe the profiles of areas combining differently high and low levels of ES diversity and of landscape heterogeneity, concluding that mosaic landscape were not always more multifunctional than homogeneous ones, depending on their composition.

Figure 5 below summarizes the framework that guided this analysis as well as the main resulting outputs.

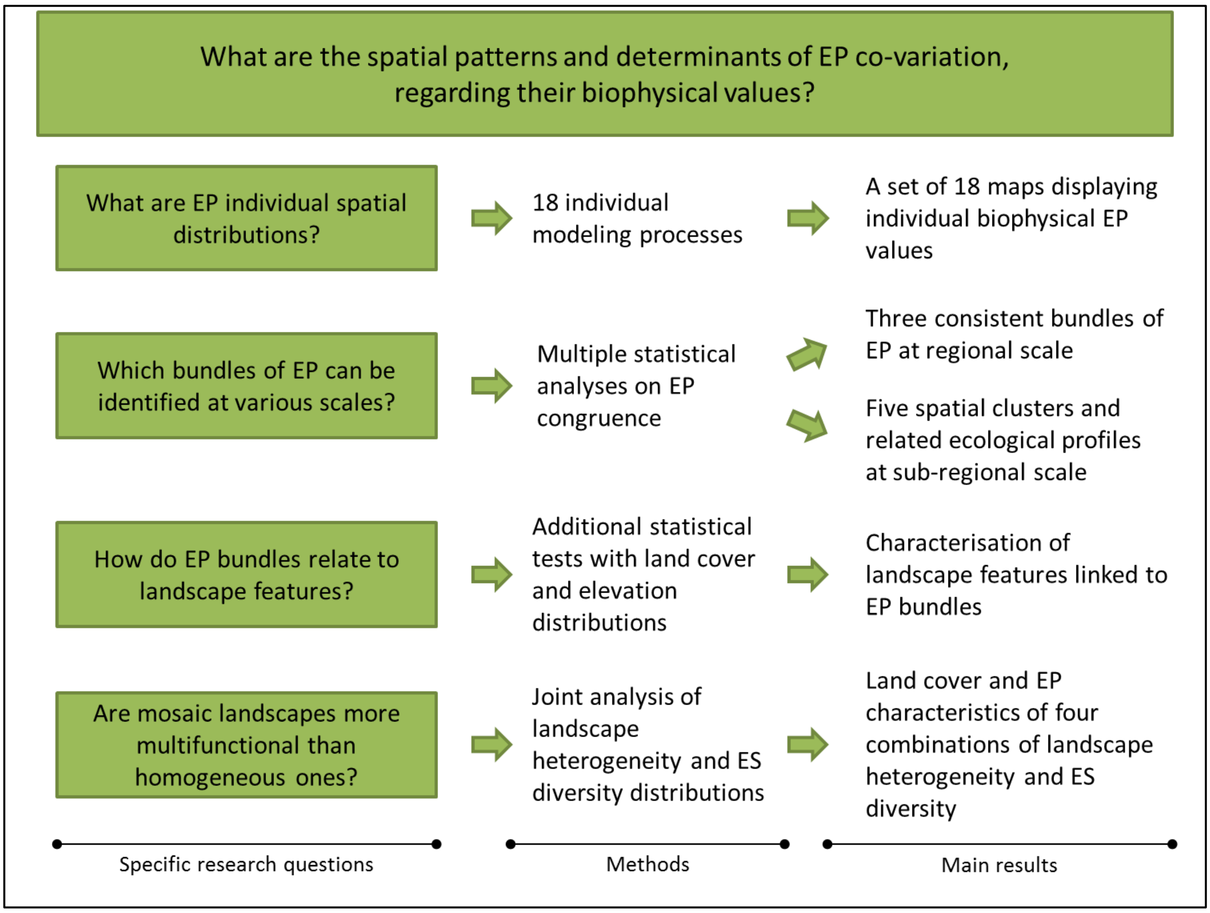


Figure 5: Specific research questions explored in the biophysical assessment of EP bundles (Chapter I), related methods and main results obtained.

Through this biophysical perspective on ecological parameter associations at nested scales, I explored general patterns and determinants of ES and biodiversity bundles depending on their spatial distributions. I contend that this work could support the governance of environmental resources, as it addressed the call for a better ecological understanding (Kremen 2005). Indeed, by applying multiple analyses to different scales, we could feed the diverse objectives pursued by governance instruments relative to policy (general frameworks and directives), planning (regional strategic plans and specific guidelines) and management (locally-tailored actions relevant for specific issues).

First, policy making could benefit from sound results on regional associations between ecological parameters as these would ease the design of reachable objectives for natural resources governance. For instance, Pearson correlation analyses concluded on multiple positive pairwise associations between forest-related ES, confirming their multifunctional role. Such results could be used as supporting rationale for general policy orientations, as those promoted in a recent report on the future of forestry by the European Commission (European Commission 2013). Moreover, in the French Alps, insights on ES relationships (e.g. the negative correlation between nature tourism and plant diversity) could reinforce policy orientations of the Alpine Convention (SPCA 1991) or of the territorial directive for Northern Alps planning (Préfecture de région Rhône-Alpes 2010). One limitation of correlation analyses is that they leave causal relationships out of scope, requiring careful application based on expert interpretation. Additional insights on relationships between EP were found during the qualitative assessment of the alpine social ecological system presented in Chapter II.

Second, adding a spatial dimension to the identification of bundles of ecological parameters enabled addressing the needs of more specific governance instruments dedicated to planning. For example, forestry regional strategic plans (e.g. ORF Rhône-Alpes 1999 for public forests and SRGS PACA 2005 for private forests) already recommend that forestry incorporate as joint objectives carbon storage, protection against rock falls and mitigation of water flows, as supported by our pairwise overlap results at regional scale. Likewise, the observed but potentially negative overlap between crop production and regulation of water quality is integrated by regional planning for sustainable farming (e.g. DRAAF 2012 at regional scale in France and UK DEFRA 2014 for a British example). Therefore, our analyses could support existing planning instruments and also help addressing new challenges seldom targeted until now (e.g. the spatial congruence in grasslands of fodder production and of the ability to regulate water quantity). In addition, the use of self-organising maps to identify clusters and the description of the ecological profiles linked appeared a very suitable tool for increased appropriation of ecological relations by society and decision-makers. Indeed, when I had the opportunity to present those results outside the scientific community, during a general public conference (*Université des Alpes*, Megève, 2013) or with stakeholders of various profiles (*steering committee of ICARE project*, see Chapter IV and general discussion), they were easily understood and their transferability for local participative land management was highly discussed. Their suitability for communication and decision-making was underpinned by stakeholders implied in land planning, conditional to in-depth analyses for specific areas.

Third, local analyses linking landscape patterns to bundles of ecological parameters appeared insightful for management of natural resources. Indeed, the overlap we found of fodder production and recreation potential with grasslands and pastures highlighted the importance of maintaining open landscapes with extensive agriculture as an indicator of naturalness and recreational attractiveness. This is already taken into account by several municipalities which subsidise livestock grazing and mowing by young farmers (e.g. issue addressed by Grenoble metropolis Agricultural and Rural Development Strategic Project for 2010/2016). Other associations not yet included in management strategies would gain in being made explicit to local decision-makers. For instance, we confirmed the relevance of productive forests and grasslands for hydro-energy production but, to our knowledge, vegetal cover is not yet incorporated into watershed management in the French Alps.

Our analysis did not quantify uncertainty but discussed the limits of our results for practical implementation (see paper in section V). In particular, we warn against the confusion between correlation and causal relationships, as while we were able to quantify correlations and spatial congruence, we did not explore causality. The use of generalised models, canonical analyses or structural equation modelling (see details in Mouchet et al. 2014) could help progressing in the understanding of driving forces and thus of causality. This would be required to limit unexpected effects of policies and management choices, as we further explore in Chapter III with the rebound-effect analysis. However, communicating on uncertainty with stakeholders remains challenging, in particular for governance and management choices where limited time-resources can be dedicated to the understanding of methods, results and implementation opportunities. The second point of attention highlighted by our biophysical analysis concerned scale issues. Indeed, when we presented our results to stakeholders, they were tempted to focus on a specific location, i.e. to interpret them at very fine scale, even though we insisted on their relevance for regional scale understanding only. How to present spatial data without risking their overinterpretation remains an open question for me. One option to limit this risk could be to map rougher shapes over areas of overall similar values instead of distinguishing between pixels that can be looked at individually.

Overall, we proposed a pattern-based approach of multifunctionality. It has the potential to raise awareness for environmental resource management at the massif scale and to open the way for more local and planning-orientated work. Many methodological issues and modelling concerns were explored during this alpine assessment and could be transposed in a research-action perspective. Moreover, scenarios could explore potential future trajectories depending on climate change, land allocation and management choices.

This biophysical assessment proposed a multi-layered description of alpine ecosystems, multiple in terms of variables, scales and associations (between ES, with biodiversity, with land cover…) considered. In particular, we stressed the interest of considering bundles of ecological parameters for environmental management. I believe this is required to anticipate the trade-offs that appear both between ES and between ES and biodiversity. Moreover this ‘bundle’ approach calls for a management at landscape scale that appears promising. Alpine regions have begun considering land planning following a landscape perspective (e.g. DIREN RA 2005). By going beyond sectoral approaches, these works rely on multiple indicators and address multiple objectives over the same areas. This trend challenges the promotion of aggregated indices (whatever the value-domains) to ease understanding and integration of environmental issues notably by policy-makers (Paracchini et al. 2011). I acknowledge that the integration of ecological features expressed as multiple biophysical values remains challenging but I also trust the ability of stakeholders to deal with more than one indicator at a time, even expressed in biophysical terms. Indeed, as expressed by Smith et al. 2011: “Possible progress on alternatives will only succeed […] challenging the idea that people cannot cope with more than one number.”

In conclusion, I promote biophysical assessments as one of the essential layers required to get a comprehensive view on social-ecological systems (van der Biest et al. 2014) and stress its complementarity with social or economic assessments, that should not be used as single substitutes (Gómez-Baggethun et al. 2011, Kallis et al. 2013, Martín-López et al. 2014).

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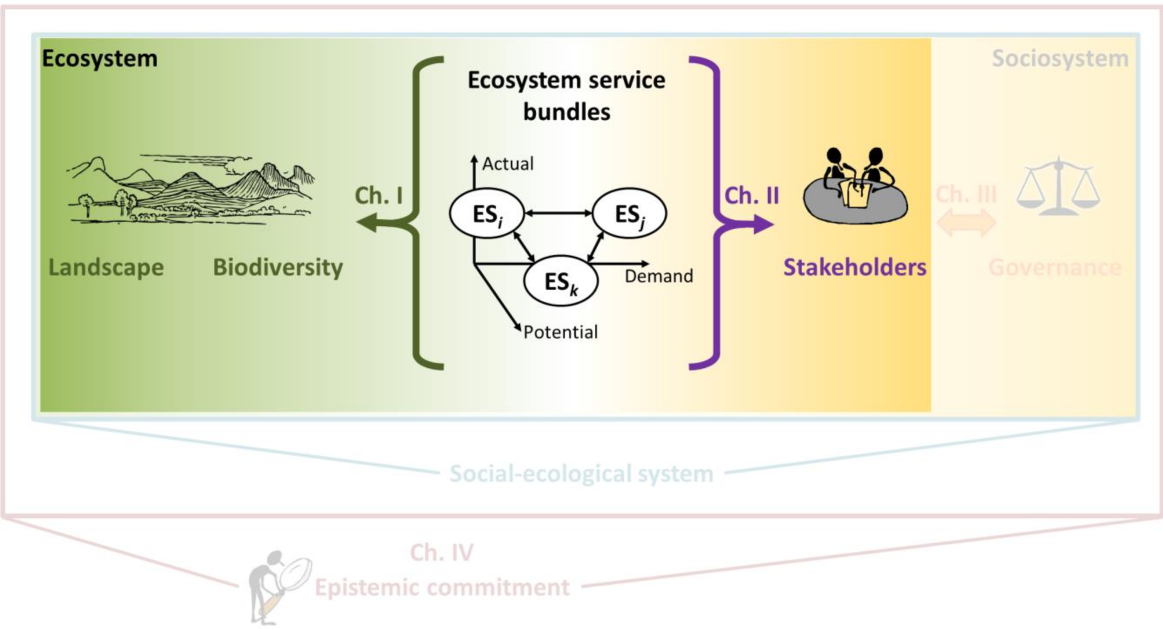
Chapter II – Influence networks

Chapter II – Qualitative analysis of influence networks



Chapter 2 – Qualitative analysis of influence networks

In Chapter I, we assessed interactions among ecosystem services and biodiversity through a pattern-based approach of their bundles, expressed as biophysical values. In order to address the social dimension of these relationships, Chapter II aims at exploring influence networks of ecological parameters (EP), defined as both ecosystem services (ES) and biodiversity variables. This was done through the qualitative analysis of a consultative process carried out with local stakeholders.

Chapter II is structured in five sections:

- Section I presents the **specific research questions** related to our qualitative analysis of influence networks around ecological parameters.
- Section II proposes **innovative methodological propositions** for EP assessments in social-ecological systems, structured as the Influence Network Framework (INF).
- Section III exposes and discusses the **four-step consultative process** we performed to explore EP influence networks perceived by local stakeholders.
- Section IV is a **paper**, submitted to the journal *Ecology and Society*, that incorporates a presentation and discussion of our **main results** regarding methodological insights and actual implementation (pages highlighted by a black border).
- Section V concludes by a **synthesis** of our main insights from this qualitative assessment of EP influence networks and discusses the methodology adopted for the consultative process and related data treatments.

I. Specific research questions

The overarching objective of this chapter is to explore how ES, biodiversity and external variables interact in the complex social-ecological system of the French Alps. I approached this objective with two questions:

1) How can influence relationships concerning ES and biodiversity be described to inform their management?
2) How is the French Alps social-ecological system perceived by stakeholders? With which implications?

To answer these questions, a consultative process was carried out with stakeholders of regional expertise to provide material for conceptualising and implementing the methodological innovations that structured our analysis.

II. An innovative Influence Network Framework (INF)

In my attempts to explore interactions within the French Alps socio-ecosystem I was confronted to a complex conceptual landscape comprising a number of recently developed frameworks and concepts that appeared insufficiently interconnected to date. So as to produce knowledge relevant for an 'ecosystem-based management' (Chan et al. 2012), we needed a framework that would explicitly capture trade-offs and synergies among ecological parameters (Rodríguez et al. 2006, Kareiva et al. 2007, Luck et al. 2012) and that could consider equally social and ecological aspects (Spangenberg et al. 2014). In this endeavour, we considered two conceptual areas.

- On the one hand, different proposals have been made to formalise interactions between ES. Bennett et al. (2009) proposed a framework distinguishing direct relations between ES from indirect relations linked to external factors. Rives et al. (2012) adapted this framework by explicitly distinguishing interactions arising from the ecosystem from those linked to the social system. Kandziora et al. (2013) proposed direct interrelation matrices to describe main supporting, reducing and feedback links between pairs of ES.
- On the other hand, ES have been described according to three distinct facets that together enable their complete understanding, and thus conditionally management (Burkhard et al. 2012, Villamagna et al. 2013, Bagstad et al. 2014). As no consensus has yet been reached on exact terminology (see dedicated paper in section V for alternative terminologies and references), they will be hereafter referred to as follows:
    - The potential supply facet represents ecosystem potential "capacity to supply services" (Bastian et al. 2012), considering its geophysical and ecological characteristics in the current land cover matrix, but notwithstanding social factors (e.g. demand, uses, economic constraints…).
    - The demand facet represents "the amount of service desired by society" (Villamagna et al. 2013), irrespective to the ability of the ecosystem to fulfill this desire. The demand facet can incorporate multiple and potentially contrasted opinions on desirable levels of ES, due to the various priorities held by stakeholders regarding environmental management (Lamarque et al. 2011).
    - The actual supply facet corresponds to the actual encounter of demand and potential supply and also includes the influence of external drivers as legislation or economic constraints.

Overall, these two conceptual areas, respectively interaction frameworks and ES facets, have evolved mostly separately and we hypothesized considering them jointly would advance the understanding of ES interactions. The innovative framework we proposed, the Influence Network Framework, sought to progress in this direction by explicitly accounting for the three ES facets in their interactions with the surrounding system, which to date had not been formalised (Figure 1).

We conceived the Influence Network Framework (INF) as a conceptual graph that creates networks of influence relationships. Its components encompass ecosystem services, biodiversity variables and external variables describing the ecological setting or social factors. These variables are connected when relevant to represent the influences they exercised on each other. The graphical output (i.e. the influence network) delivers a comprehensive overview of the social ecological system that could inform management or foster collective learning. We further detail the operationalization of the INF and discuss its characteristics in the paper presented in section IV.

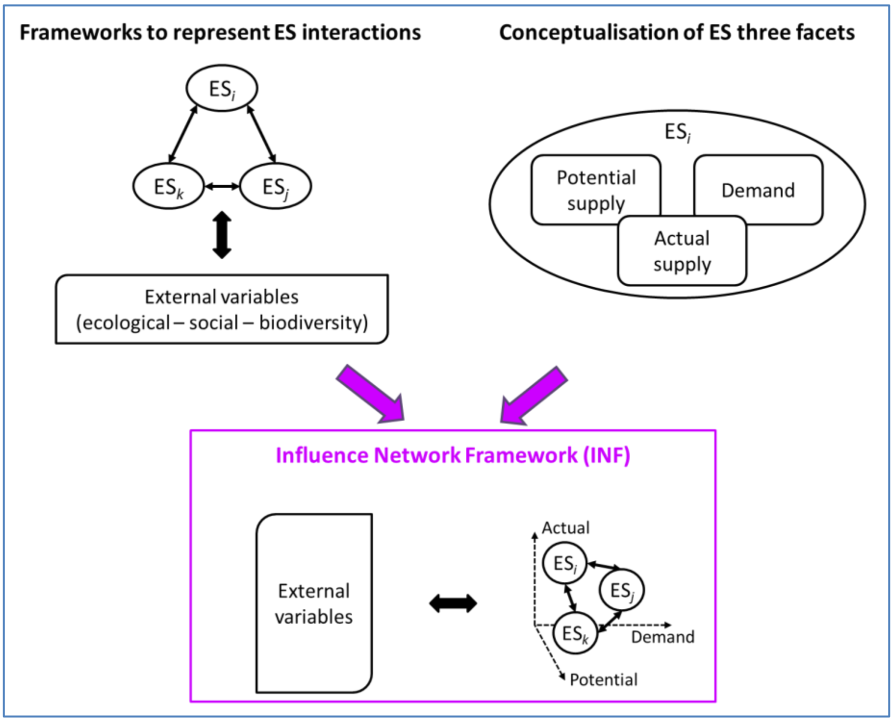


Figure 1: Conceptual origin of the Influence Network Framework (INF), at the junction between developments for representation of ES interactions and conceptual progress in ES facet description. A formal description of the INF is proposed in the dedicated paper presented in section IV.

III. A four-step consultative process

To test the operationalization of the suggested INF and to progress in the understanding of the alpine social-ecological system, we carried out a four-stepped consultative process. In the view of consistency, the initial consultation phase is hereafter referred to as "step 0" and more precisely commented below as it was not included in the paper presented in section IV. This paper focused on the three following phases (namely steps 1 to 3). Steps 1 to 3 explicitly referred to ES facets as proposed in the INF (section II) while the initial step can be seen as a general approach of the alpine territory and of its specificities. I led the whole consultative process for Steps 1 to 3 (stakeholder selection, organisation and content of the consultation, result treatments, reporting back, post hoc treatments and conclusions). Additional details on institutions and expertise of the stakeholders involved can be found in the paper presented in section IV.

A. Regarding stakeholder involvement

Involving stakeholders in so-called 'participative research' projects gained in popularity in the last decades (Menzel and Buchecker 2013, Pade-Khene et al. 2013) as it is expected to result in better effectiveness and more sustainable governance of environmental management (Palomo and Montes 2011). The European project CONNECT, within which my thesis has been developed, accordingly aimed at engaging with regional stakeholders to reinforce the environmental assessment carried out and in particular strengthen the related governance and policy analysis. Thus, we explored the questions of 'who to involve' and 'what for' regarding our French Alps assessment. Relevant stakeholders to engage with are usually defined as these "who will influence or be affected by […] actions arising from the planning process, or be responsible for implementing these actions" (Ban et al. 2013). Reasons for engaging with stakeholders are numerous, and can be usefully categorised in two (Reed et al. 2009). First, normative approaches involve stakeholders as a way to legitimize decisions that are made by empowered key actors, in particular when consensus needs to be reached and knowledge shared. Second, instrumental approaches are directed towards the understanding of relations between stakeholders and specific issues, in the objective of better managing them in an adaptive way or of preventing conflicts among stakeholders of various priorities and concerns. Moreover, various degrees of participation can be attributed to stakeholder groups (Arnstein 1969, Luyet et al. 2012).

While we acknowledge the relevance of co-decision and empowerment in research-action projects, our concerns remained more academic and less governance-orientated, leading us to focus on collecting opinions and knowledge and reporting back on general results. Thus, our approach can be described as rather instrumental as we mostly aimed at consulting stakeholders to inform our understanding of the alpine system.

Stakeholder sampling was intentional and reflected the need for "information-rich cases" (Coyne 1997, Menzel and Buchecker 2013). Following a classical case-study research approach (Eisenhardt 1989), criteria for their selection included balancing between academics and non-academic professionals, focusing on institutions with recognised competencies and adequate scope (spatially and in their objectives) and representing the various domains of competence concerned by environmental management. We used a snow-ball sampling strategy initiated by consultation with scientific partners and previous non-academic collaborators. Our sampling does not claim exhaustiveness, as we focused on regional representatives from recognised institutions only, however I believe that this sampling successfully informed our description of influence networks among ecological parameters. An interesting follow-up would consist in exploring variations in opinions, priorities and concerns between stakeholder groups so as to use the INF as a communication and collective learning tool useful for sustainable environmental management (Lamarque et al. 2011). Moreover, additional stakeholder categories could be integrated to account more for individual concerns (e.g. tourists, shepherds, residents…) or supra-regional priorities (i.e. to connect a regional assessment to surrounding issues and governance instruments).

An overview of the profiles of stakeholders consulted is proposed in the paper presented in section IV (Figure 4).

Choice of participative techniques has been described as depending on various factors, including degree of stakeholder involvement, type of stakeholders, context of the process, timing and economic constraints, and facilitation skills (Luyet et al. 2012). Finally, we used three different techniques to collect information. We expose them and discuss their interests and limits along with the general description of the four steps of our consultation.

B. Step 0: framing the context

1. Methods

Collaboration with scientists from Alterra - Wageningen University & Research Centre created the opportunity for a common workshop in November 2012. This meeting was included in VOLANTE (http://www.volante-project.eu/), a broader research project dealing with '*Visions of land use transitions in Europe*'. Part of VOLANTE project aimed at comparing the relevant driving forces in land use change for different areas over Europe, leading to four workshops in contrasted areas: continental regions (Romania), mountain regions (Alps), Atlantic zone (Denmark) and Mediterranean zone (Greece).

I was involved only in the meeting that concerned mountain regions, for which I was responsible of the selection and the pre-workshop dialog with stakeholders and of facilitation during the workshop. Moreover, I was thoroughly involved in the data analysis and interpretation, and in reporting back to stakeholders.

Our aim with this specific workshop was twofold: first, to understand the changes in landscapes and land uses in the French Alps during the last 25 years, and second, to clarify the main driving forces responsible for these recent trends. Driving forces are the forces that cause observed landscape changes, i.e. these influencing the trajectories of landscape development (Bürgi et al. 2004). They can originate from various domains: political, economic, cultural, technological and natural driving forces are usually distinguished. Moreover, they emerge and operate at different scales, from international to local. Identification of driving forces for land use change is a useful step for understanding and managing the dynamics of landscapes and their resources in complex systems (Hersperger & Bürgi 2009). Given this objective, we invited nine stakeholders with regional expertise in natural resource management for a one-day focus group and proposed to deliberate using a specific participative method, called Fuzzy Cognitive Mapping (hereafter FCM refers to fuzzy cognitive map).

FCM are a graphical representation of a complex system where i) driving forces influencing the core problem are displayed, and ii) influence relationships between them are symbolized. More formally, fuzzy cognitive mapping is a method to approach system dynamics, i.e. "the behaviour of complex systems over time" (Kok 2009). A FCM is built in two steps. First, stakeholders individually identify driving forces and then collectively discuss them until consensus is reached on their precise meaning. Second, stakeholders jointly assess the strength with which each force is perceived to be connected to others and to the core issue, land use change in this case. A *post hoc* treatment of the FCM obtained consists in quantifying temporal changes of the system, based on the relative value of all influence relationships (-1 to +1). Further, the importance of all driving forces is defined by an 'initial state vector' that describes the initial setting (0 to 1). Then, the initial state vector is modified by successive runs implementing the resulting influence of all relationships (Kok 2009). The final output is a graph showing the trends for the core issue (here land use change) and driving forces over time. While the initial state vector and the values describing influences rely on past trends (here the last 25 years), the graph output represents a projection of the potential changes in the system in the future.

2. Main results

As they are not part of the paper presented in Section IV, I briefly present hereafter the main results of this workshop, while for the other steps, results will be described in the paper only.

The sequence of results from the FCM process is represented in Figure 2 and characteristics of driving forces in Figure 3. Direct outputs from the workshop were not directly usable by the dynamic simulation model that required simplified feedback effects. In order to focus on overriding forces, we needed to simplify interactions. For this we merged closely related driving forces. Finally, eleven driving forces were collectively identified and defined to explain recent changes in French Alps land uses (Table 1).

Table 1: Driving forces identified and defined in the focus group as prominent in landscape change in the French Alps for the last 25 years.

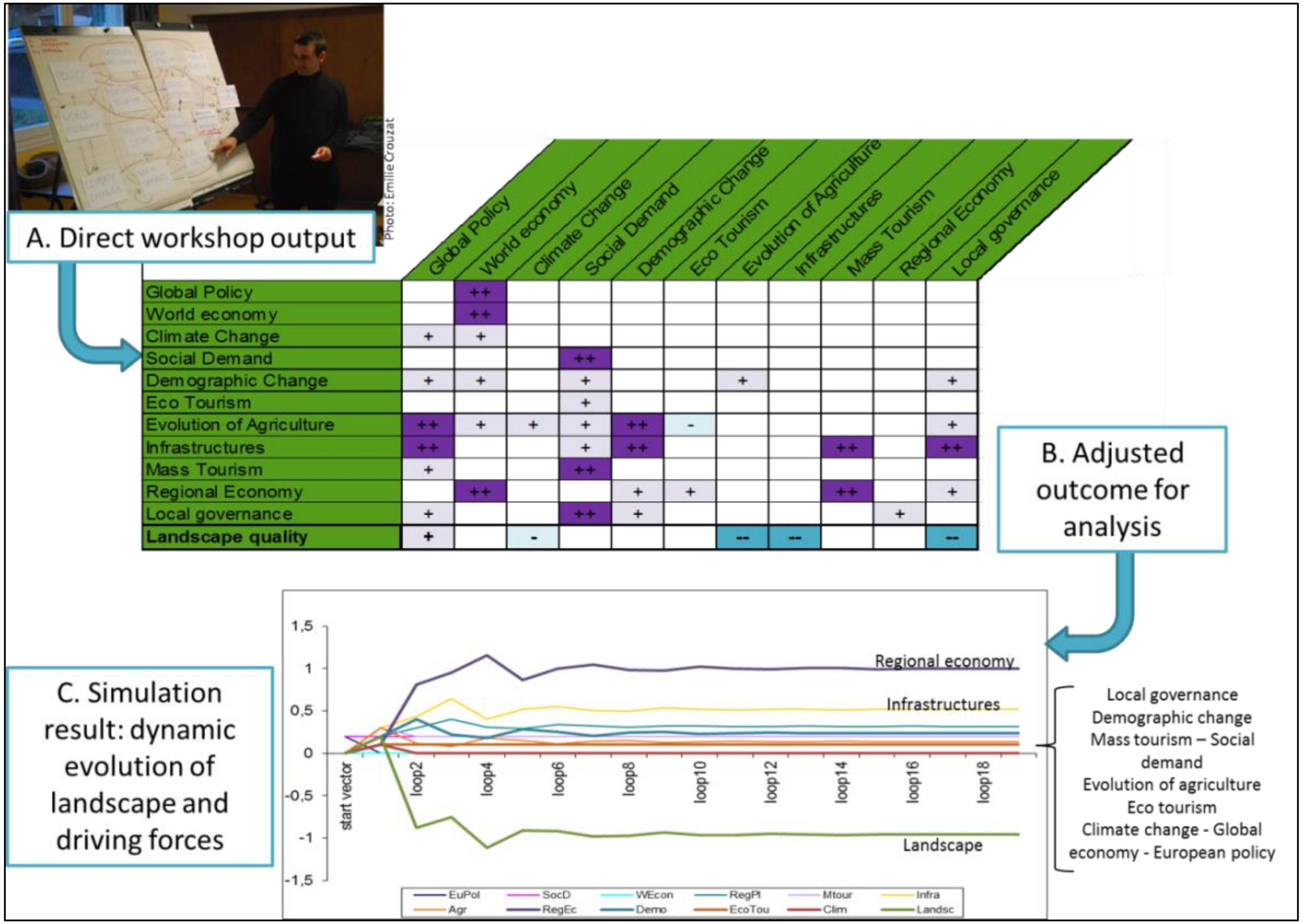
Driving force	Consensual description by stakeholders
Global policy	Global governance setting represents the influence of regulating and incentive policy instruments defined at European and national scales. Main instruments considered include the European **Common Agricultural Policy** as it is assumed to have a major influence on agricultural land use; the **Natura 2000** network for its widespread influence over the territory; **Lisbon treaty** due to the special recognition of mountain areas within the structural organisation of the European Union; as well as different **cross-border cooperation programs** specifically addressing alpine issues. These tools follow a **hierarchy** of influence.
Local governance	Local policy tools represent the local version of global orientations that are adapted to local conditions and stakeholders, and complemented by local traditions and rules. Some sectors are strictly controlled, like **waste and water management** or **protection against natural hazards**. **Urban pressure** in the Alps is very strong, with an intense peripheral urbanisation surrounding a more preserved core mountain area. **Planning** is perceived as focused on urban areas and not planned at supra-communal level, leading to a **lack of coherence and efficiency** in land use and resources management especially in areas composed by many small independent municipalities. The main policy instruments discussed included the '**Loi Montagne**' as a specific law for urban development in mountain areas and **spatial protection status** (e.g. regional natural parks) for their contribution to territorial specialisation.
World economy	World economy is affected by **market globalization** and **internationalization of investments**. The balance of trade between **imports and exports** evolved in the recent past, with sectorial specificities (e.g. under-exploitation of alpine forests due to Northern European countries competiveness, opening to global food markets in agriculture).
Regional economy	**The maintenance and creation of jobs** is a critical key factor to maintain populations and land uses. The Alps are characterised by a strong **contradiction** between the need of economic activities and territorial land use planning coherent with ecosystem sensitivity. The influence of the **building** and **energy** sectors are highlighted.
Climate change	Climate change impacts are both **direct** (on ecosystems) and **indirect** (on practices). **Impacts on alpine ecosystems** are due for example to glaciers melting, variation of hydrologic regimes or migration in plant distributions. **Management and production practices in the agricultural, forestry and tourism sectors are currently adapting**, even if the timespan considered here is short relative to these changes and only represents the initiation of transformations to come.
Social demand	**Private property** is culturally highly important and is translated in the diversity of management options chosen by land owners, even within the current governance system. In addition, individual choices in terms of **consumption, activities and housing** convey a certain type of **land use demand** and of **relationship to nature**.
Demographic change	Population characteristics in the French Alps are linked both to **demographic heritage** reflecting regional **attractiveness** and constraints, with contrasting features for Northern and Southern Alps, and to **current migration trends** characterised by widespread pendulum migrations (e.g. France – Switzerland commuters) impacting infrastructures and social life.
Infrastructures	The **accessibility** of the Alpine territory is highly dependent on **transport** infrastructures, which deeply impact **ecosystem fragmentation**. **Energy costs** are key factors in population mobility and are recently becoming limiting. The development of new infrastructures and the future of existing ones is sometimes perceived as **disconnected from regional land plans** due to local informal arrangements.
Evolution of agriculture	Agriculture has been lately subjected to many changes leading to changes in zonation and intensiveness of practises. **Mechanization** led to hillsides and wetlands **abandonment** in favour of the **intensification** of more accessible and productive lands. **Sub-urban production** intensifies, in response to a higher demand for local products and the development of farm-to-fork processes. **Pastoralism**, characteristic of mountain areas, also evolves: grazed areas are nowadays concentrated in valleys and high altitude meadows only. This absence of grazing pressure favours the appearance of a woody intermediate layer. Local practises and farmer's income are enhanced by **quality labels and certificates** (e.g. AOP, IGP).
Mass tourism	Mass tourism, in **winter** overall, is seen as "**invasive**" and very impacting on landscapes (activities, housing and transportation). Its **economic spill overs** are essential for numerous inhabitants in the Alps and create **financial transactions** through building investments and individuals' placements. Municipalities can choose various management practises with

	different impacts on urbanism and land uses which are characteristic of different **touristic development models**.
Ecotourism	Scattered tourism represents a « **smoother** » relation to nature, more **diverse** than mass tourism in terms of practise types (hiking, biking, farm's visit…) and seasonality. It creates **income** for rural inhabitants and allows the **promotion of traditional landscapes and typical architecture**. This kind of tourism also affects **urbanism schemes** through a high demand of **second individual housing** impacting spatial structures of alpine municipalities.

The FCM projected a future negative trend for landscape quality over the French Alps whereas the regional economy and infrastructures were projected to improve (Fig. 2.C). The trend in landscape change was mainly driven by three factors: policy originating from the European Union, social demand and world economy. These factors were highly influential on the whole FCM, while receiving limited influence from other driving forces (Fig. 3.A).

Main negative forces on landscape quality were evolution of agriculture, climate change, infrastructures, mass tourism, local governance and global policy (Fig. 2.B). Negative influences of climate change and of infrastructures were straightforward due to the induced additional constraints and artificialization of landscapes, respectively. Evolution of agriculture (i.e. "intensification in favourable areas and abandonment of the naturally disadvantaged areas") was mentioned as negative for landscape and biodiversity and linked to demographic changes. This was exemplified by the situation of Southern Alps where declining attractiveness of agriculture led to declining population, in particular from the agricultural sector, explaining the decreased management of landscapes and the resulting colonization by forest and shrublands. Meanwhile, ecotourism favoured extensive agriculture maintaining cultural landscapes by inducing higher income to local farmers selling high added-value products. Thus ecotourism was mentioned as negative for the 'evolution of agriculture' force. At the opposite, mass tourism increased infrastructure and its negative effects. The development of this activity reinforced the priority given to regional economy rather than to landscape quality. Moreover, the present local governance system was mentioned as negative due to the perceived lack of consistency in planning across municipalities. Strong influence from the regional economy was also mentioned as threatening landscape quality through its lobbying capacity on local decision-makers. Global policy and social demand were negative as they reinforced various negative drivers.

Stakeholders collectively attributed a varying importance to the different driving forces (Fig. 3). This information is policy relevant as it can enable prioritising actions to limit land use changes. A strategy for maintaining landscape quality could be to focus on targeting highly impacted forces. Indeed, they are influenced by numerous other driving forces which could be targeted by multiple management measures so as to weigh on landscape quality. For instance, agriculture is highly impacted by other driving forces while its changes directly influence landscape quality. Thus, influencing the drivers of agricultural changes could support extensive farming and its contribution to promoting high quality landscapes. Indeed, numerous policy instruments already exist that aim at supporting extensive agriculture (agro-environmental measures from the Common Agricultural Policy, development of geographical indications for high added-value products, etc. See also Chapter III). An alternative solution would focus on highly influential drivers, namely social demand, global policy and world economy. However, we believe these drivers to be actually out of reach for alpine decision-makers and rather consider them as external and quasi fixed constraints, i.e. as boundary conditions. The intermediate position of local governance (Fig. 3 A and B) indicated its particular relevance for maintaining landscape quality as an adequately flexible driver at appropriate spatial and temporal scales.



	Global Policy	World economy	Climate Change	Social Demand	Demographic Change	Eco Tourism	Evolution of Agriculture	Infrastructures	Mass Tourism	Regional Economy	Local governance
Global Policy		++									
World economy		++									
Climate Change	+	+									
Social Demand				++							
Demographic Change	+	+		+			+				+
Eco Tourism				+							
Evolution of Agriculture	++	+	+	+	++	-					+
Infrastructures	++			+	++				++		++
Mass Tourism	+			++							
Regional Economy		++			+	+			++		+
Local governance	+			++	+					+	
Landscape quality	+		-				--	--			--

Figure 2: Results of the FCM process: A. direct outputs from the workshop, B. adjusted outcomes for analysis, C. results of the dynamic simulation

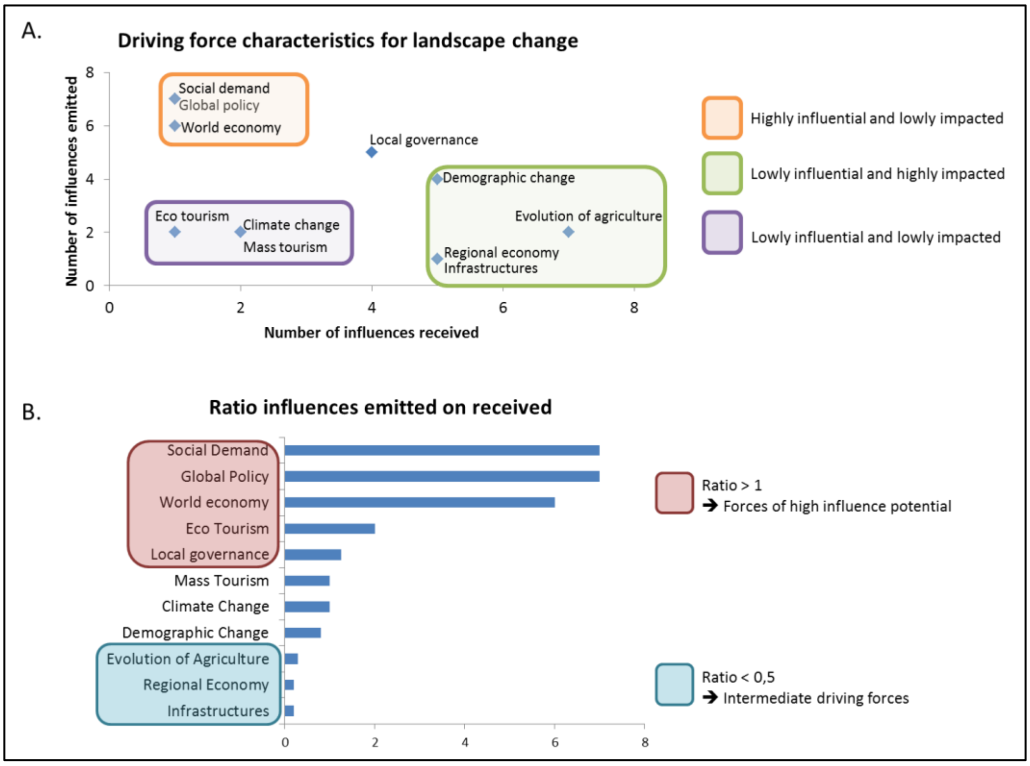


Figure 3: Characteristics of driving forces responsible for land use change according to stakeholders involved in Step 0. A. Scatter plot representing the number of received influences as a function of the number of emitted influences. B. Ratio between influences received and emitted.

3. Discussion

In our experience, the cognitive mapping method appeared relevant for engaging discussion among stakeholders of various backgrounds. Through iterative discussions, they collectively proposed consensual definitions of the driving forces and further agreed on the importance and sign of the influences linking driving forces among them and to landscape quality. However, if strongly divergent opinions are expressed by stakeholders, I am not sure whether this collaborative method could help overcoming them. Good facilitation skills are required to ensure equitable allocation of speaking time as well as to adequately transform stakeholders' narratives into FCM elements.

The FCM demonstrated its potential for collectively producing a comprehensive and dynamic view of driving forces influencing land use change. One main interest is its ability to deal with internal feedback loops, stocks and flows so as to get a more comprehensive view of potential nonlinear behaviour of systems. A second main advantage holds in its position in between quantitative and qualitative methods. As strengths of influences are appreciated in a semi-quantitative way and relatively to each other, FCM can be adequately used for connecting workshop results with models and thus better incorporates stakeholder inputs (van Vliet et al. 2010). However, its main drawbacks related the complexity of dealing with highly interconnected driving forces that could get confusing during the workshop, as well as the need to adapt workshop outputs to requirements of the dynamic model (leading to their *a posteriori* simplification). Moreover, the necessity to positively and negatively weight influences was problematic for some stakeholders as they preferred weighting the strength of influence in absolute terms, relative to each other, but were reluctant to judge it as positive or negative. For instance, the influence of social demand on landscape was ascertained but telling whether it affected positively or negatively its quality was not straightforward as it implies a subjective judgment on what makes a 'nice' landscape (which moreover remains a pure social construction). Being very clear on the common definition of driving forces allowed us overcoming this issue by getting more objective on the influence discussed (by specifying that the sign of influence was not a personal judgment but a codification relative to actual trends).

Regarding the outputs of the cognitive mapping, I have some concerns about result interpretation. First, FCM relies on the hypothesis that changes in landscape quality can be understood by the sum of pressures from individual driving forces, and that positive forces can compensate for negative ones. Complex synergistic and antagonistic effects between forces are therefore accounted for in an integrative manner as a whole, i.e. without a clear attention to individual effects of driving forces. I wonder to which extent the mathematical calculation using state vectors and relative influence values can represent reality (i.e. I am not sure that positive and negative forces can actually compensate each other effects). Second, it is not clear from my experience to which extent the list of driving forces and their influences were conditioned by the opinions and dynamics represented within the specific group of stakeholders we consulted (small group size), i.e. to which extent our results could be generalised. This might however not be a real concern if users of the FCM outputs are clearly aware of what is actually represented by the results, i.e. a subjective vision of interactions as depicted by a group of individuals of various backgrounds. However, applicability of the outputs, e.g. for governance purposes, dramatically decreases if reliability of the map cannot be soundly assessed. Overall, disentangling causal factors remains challenging and I support the calls for a “portfolio approach to understanding socio-ecological systems” (Young et al. 2006) that would combine several methods to approach the systems assessed sequentially. Indeed, convergence of results from two or more methods would increase confidence in the results while contradictory results should lead to additional analyses.

C. Step 1: Setting the stage for the INF assessment

After framing the general context of recent landscape change, our consultative process focused more precisely on social impacts on natural and managed ecosystems. We explored how ecosystems are specifically used, conserved or impacted by the four sectors of activity that happen to be mostly responsible for their changes: agriculture, forestry, tourism and urbanism. Two questions structured this investigation:

- What **demands** are expressed regarding ES and biodiversity?
- What **actual use** is made of ES, and with which **impacts** on biodiversity?

To answer them, I carried out eight individual semi-structured interviews with regional experts, balancing between academics and socio-professionals from institutions with recognised competencies and adequate scope (e.g. the environment officer from the national syndicate of ski resorts for the assessment of the tourism sector, the head of the agricultural department of the regional government for the assessment of agricultural sector).

Semi-structured interviews were chosen in this first step and also in the third one of the consultative process as they are known to provide “reliable, comparable qualitative data to get a practicable understanding of stakeholders’ knowledge, intentions and actions” (Lugnot and Martin 2013). We extracted much valuable and relevant information as the flexibility of the interview structure enabled in-depth insights specific to the domain(s) of competence of each stakeholder. Main drawback of these interviews related to their highly time-consuming implementation (individual interviews) and treatment (transcription, coding and merging of all interviews following a deductive qualitative content analysis process (as detailled in Elo and Kyngäs 2008, Lugnot and Martin 2013). Moreover, semi-structured interviews were not iterated and consequently we could not directly confront stakeholder opinions on conflictual or uncertain issues (Reed et al. 2009).

The interview template used is presented in Figure 4, and included four open questions.

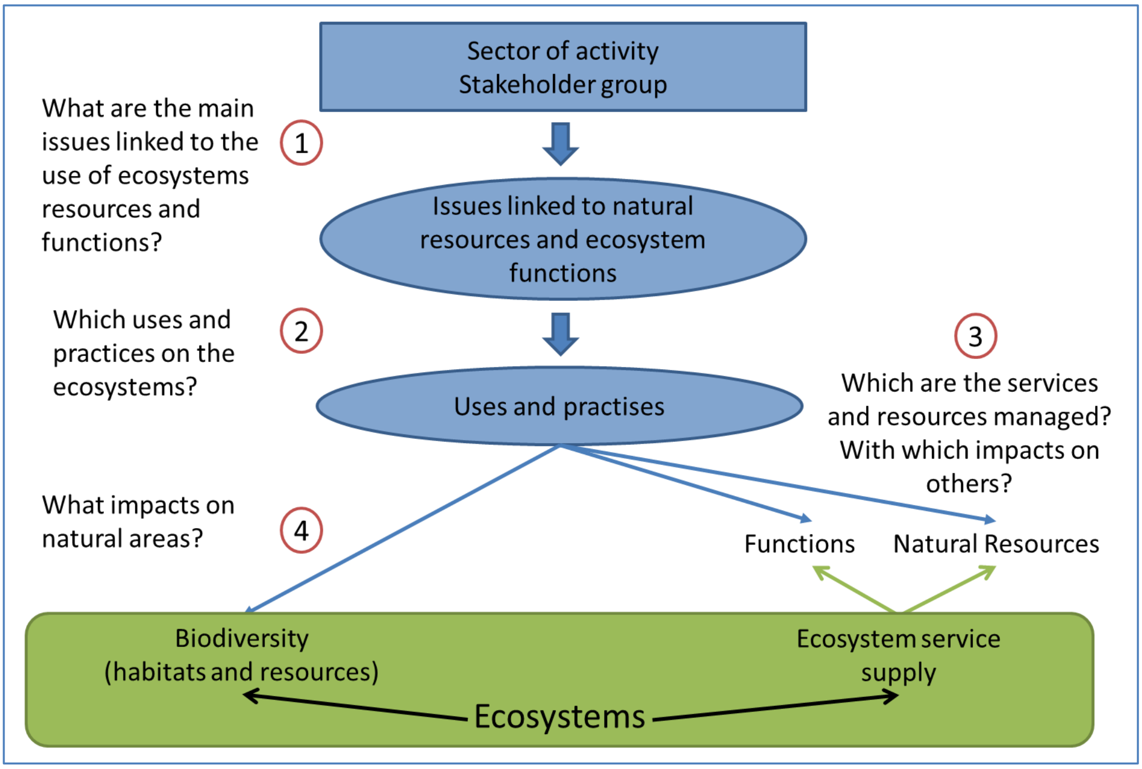


Figure 4: Template grid for the semi-structured interviews of Step 1 of the consultative process. Step 1 explored how ecosystems were specifically used, conserved or impacted by the four sectors of activity that happen to be mostly responsible for their evolution: agriculture, forestry, tourism and urbanism.

The main results from this consultation consisted in four sectoral syntheses following the template proposed in Figure 4. We identified the current uses and practises on alpine ecosystems that respond to main development issues faced by each sector of activity (Questions 1 and 2). From these, we expanded on the list of ES set as management targets and these impacted as side-effects (Question 3). Particular attention was given to general consideration of and impacts on biodiversity (Question 4). Synthetic sectoral schemes are available as at the end of the manuscript in the Appendices from Chapter II (Section A in French).

Additionally, I used this opportunity to ask about main policy instruments relevant for the management of issues discussed, which will be thoroughly explored in Chapter III on governance analysis.

D. Step 2: Exchanging views

Our analysis proceeded with the exploration of main synergies and trade-offs among ES and biodiversity in the French Alps, due to environmental influencing variables and interactions between stakeholders. We specifically aimed at addressing two questions:

- What are the important **positive and conflictual interactions** among biodiversity and ES, respective to their three facets?
- In an alpine context, which **generic influence relationships** do stakeholders perceive between ES, biodiversity and external variables?

Answers were provided by a one-day focus group gathering fifteen attendants, selected with the same requirements than for step 1. Successive sessions were conducted to focus on issues specific to the following landscapes: a) forested areas, b) agricultural landscapes and open (semi-) natural spaces, and c) artificial areas (including urban areas, ski resorts and infrastructures).

This focus group allowed for additional insights through discussions between stakeholders of varying concerns and priorities. Collective brainstorming during specifically orientated sessions (e.g. on ES networks within alpine forested areas) led to rapid understanding of complex situation involving stakeholders of contrasted priorities. Outputs were easily treated as participants collectively designed consensual and synthetic answers on the issues discussed. However, preparation time ahead of the focus group was high and we could not explore thoroughly all influence networks due to time issues during the focus group, highlighting the complementarity of this technique with semi-structured interviews. Overall, as mentioned for FCM previously, good facilitation skills are required to avoid domination of certain stakeholders during collective discussions. During the whole process, we were not faced with marked oppositions among stakeholders nor with conflictual or highly tensed situations. However, I acknowledge the need for academics engaging in participative methods to get prepared for such situations to happen and thus to previously develop their facilitation capacities as well as their understanding of local context and sources of disagreement.

Prior to the workshop, I had extracted from the discourses of the stakeholders consulted in Step 1 important positive and negative influence relationships among ES and biodiversity. I individually exposed them in cards that were presented at the beginning of each session during the focus group in Step 2. We asked attendants to pick the four cards they found most important or interesting to discuss collectively. Blank cards allowed them to propose additional relationships. Then, stakeholders displayed the cards they selected on a collective table representing which stakeholder groups were mostly concerned by each interaction. They collectively discussed most frequently proposed interactions. We asked stakeholders to explain the context in which each interaction took place and the reasons of its relevance for environmental management in the area. This allowed us to investigate synergies and trade-offs among ES and biodiversity as well as to assess their determinants. Figure 5 summarises our methodological design.

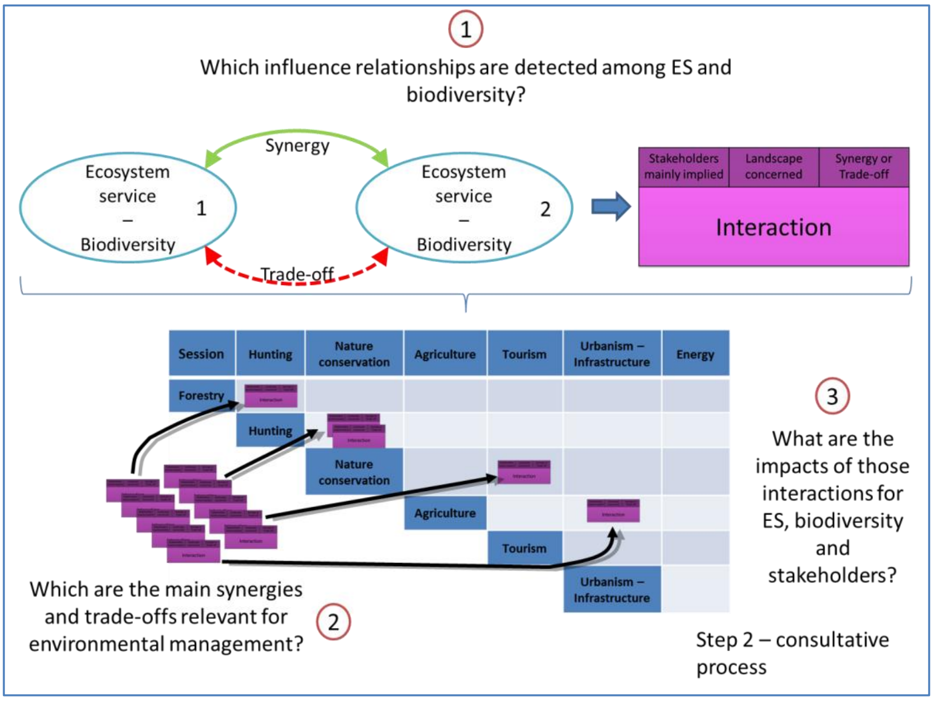


Figure 5: Methodology for the focus group of Step 2 of the consultative process. Step 2 aimed at exploring the main synergies and trade-offs among ES and biodiversity in the French Alps, due to environmental influencing variables and interactions between stakeholders

Through this process, we identified simple influence relationships among ES and biodiversity perceived as important by stakeholders. This outcome also incorporated their descriptions of influential external social and ecological variables. As post hoc treatment, we specifically attributed these relationships to ES facets and obtained a first implementation of our conceptual framework (INF) by aggregation of simple influences and related variables. Additionally, we calculated the ratio between the number of emitted influences and the number of received influences for the various categories of variables. This allowed us to approach the overall perception of the social-ecological system as discussed by stakeholders (see dedicated paper in section IV). As discussed for the FCM (step 0 above), the reliability of the results was conditioned by the set of stakeholders consulted. Indeed, additional relationships would have been provided by experts of different backgrounds. In particular, more importance could have been given to regulating services and biodiversity as a basis for the ecological functioning of the system if more expertise in ecology and environmental sciences had been integrated. Our conclusions on the general perception of the social ecological system could thereby be less distorted toward provisioning and cultural aspects which are usually more easily discussed and integrated in management concerns. However, we believe that our set of stakeholders remains close to the general perception of ecosystems by a broad public. We find these differences between perceived and actual functioning quite informative on widespread knowledge gaps that contribute to threatening a sustainable management of alpine natural resources.

An additional activity during this focus group related to the governance analysis presented in Chapter III. During the last part of the day, we tested a list of criteria proposed by CONNECT partners for assessing the environmental effectiveness of governance instruments. Our stakeholders focused on four instruments of their choice and provided us feedbacks on whether the criteria were understandable and whether information was actually available to inform them. This experience is further detailed in Chapter III.

E. Step 3: Validating and refining findings

The final step of our consultative process explicitly aimed at uncovering the influence networks of specific ES which appeared important for environmental management all along the consultation. In particular, we explored the two following questions:

- What are the main **variables influencing** the **potential supply**, the **demand** and the **actual supply** of given ES?
- What are the main **variables impacted** by the actual supply of given ES?

To complete the influence networks that previous steps approached, I performed twelve individual semi-structured interviews with regional experts selected with the same requirements than for steps 1 and 2. The methodological design of these interviews is presented in Figure 6.

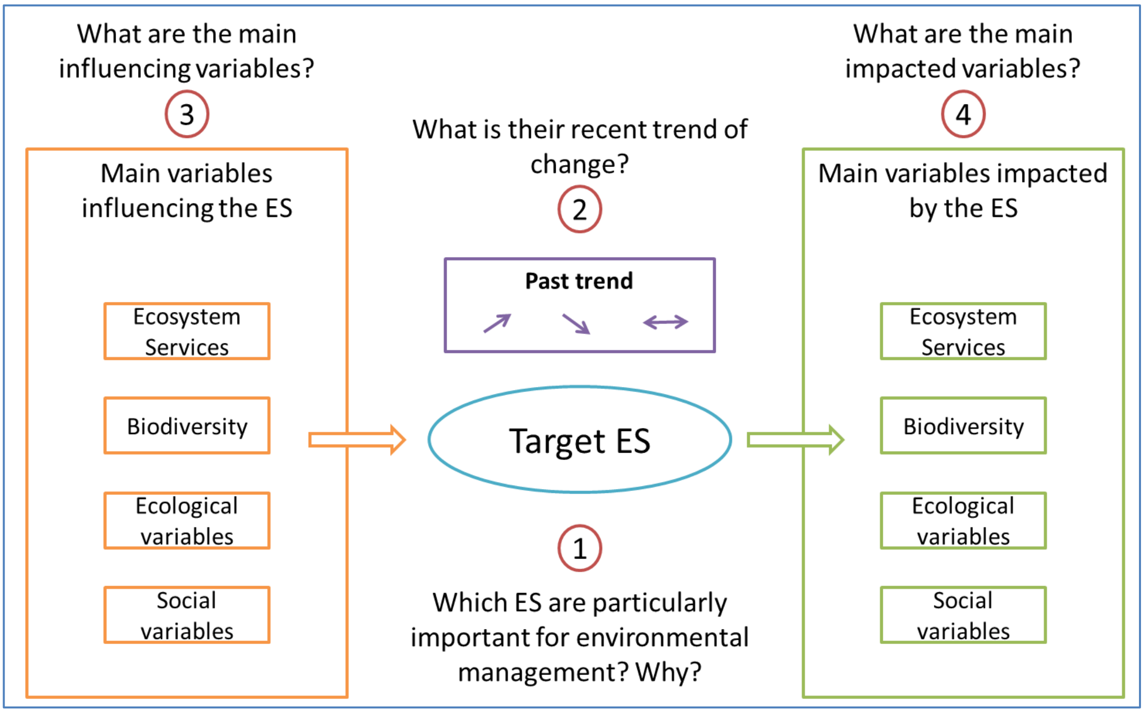


Figure 6: Template grid for the semi-structured interviews of Step 3 of the consultative process. Step 3 explicitly aimed at uncovering the influence networks of specific ES which appeared important for alpine environmental management throughout the consultative process.

Carrying out this set of interviews confirmed and completed the list of influence relationships we had gathered in previous steps and which finally reached around 200 pairwise relations. The precise description of the interactions by stakeholders allowed us to attribute them to specific ES facets as a post hoc treatment (i.e. we did not include explicitly the three ES facets in the interviews to facilitate discussion with stakeholders, and rather attributed the influences they described us to the specific facet of the ES they referred to as a latter step). Further, we confirmed the general influence sequence describing the perception of alpine social-ecological system by consolidating the ratio between emitted and received influence relationships (see dedicated paper in section IV).

F. General conclusions on the alpine system

Overall, the INF provided an increased understanding of the complex interactions among society and ecosystems across the French Alps.

Regarding pure ecological relationships between ES and additionally with biodiversity, our consultation revealed widespread gaps in common ecological knowledge. Indeed, biodiversity and regulating services were mentioned mostly as impacted variables of low influence on the overall system, i.e. of low utilitarian value regarding ecological functioning. This can be related to an actual low understanding of natural processes by many stakeholders, leading to their low consideration in management compared to social factors such as land allocation choices. Our findings were consistent with other studies where ‘visible’ services (i.e. provisioning and cultural) were more spontaneously mentioned as important by stakeholders compared to regulating ‘invisible’ services (e.g. Lamarque et al. 2011) and where the influence of stakeholder backgrounds and of local context on valuation was highlighted (e.g. Oteros-Rozas et al. 2013). Education and communication on the dependence of human societies on natural systems therefore still remain to increase and should concern a diversity of stakeholders in age, backgrounds and responsibilities.

Additionally, our results ascertained the complexity of relationships among society and ecosystems. The long-lasting shaping of landscapes, and thus of ecosystems, by human activities created cultural landscapes iconic of their mutual development. Regarding the interplay among actors, we highlighted both collaborations (e.g. co-constructed approaches to pastoralism and ski resort management) and conflicts (e.g. regarding the regulation of wild ungulate populations). These are well known by concerned stakeholders but could be highly informative for stakeholders of other domains or for decision-makers. The influence of governance choices appeared overwhelming, in a context of strong spatial and abiotic constraints on land allocation and of contrasted and yet pressing social demands within the alpine region.

Lastly, it is important to bear in mind that our assessment focused on general trends applicable at regional scale mostly. We stress its interests for academic concerns and high to intermediate-level governance institutions (i.e. down to regional level). I believe that applying the same kind of consultative process using the conceptual INF framework to structure discussions and results holds strong potential at smaller scales (e.g. community of communes) for collaborative land planning.

IV. Results - Disentangling trade-offs and synergies around ecosystem services with the Influence Network Framework – Illustration from a consultative process over the French Alps

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References from the paper are available at the end of Chapter II.

Supporting Information is available in the Appendix section at the end of the manuscript (Appendices from Chapter II (Section B)).

Abstract

Sustainability is based on maintaining ecosystem functioning while improving human well-being. For this, the ecosystem service (ES) approach has potential to bridge the still existing gap between ecological management and social development, especially by focusing on trade-offs and synergies between ES and between their beneficiaries. Several frameworks have been proposed to account for trade-offs and synergies between ES, and between ES and other components of social-ecological systems. However, to date, insufficient explicit attention has been paid to the three facets encompassed in the ES concept, namely potential supply, demand and actual supply, leading to sub-optimal descriptions of ES interactions. In this paper, we expand on previous frameworks by proposing a new Influence Network Framework (INF) based on an explicit consideration of influence relationships between these three ES facets, biodiversity and external variables. We tested its ability to provide a comprehensive view of complex social-ecological interactions around ES using a consultative process focused on environmental management in the French Alps. A synthesis of perceptions from consulted stakeholders conveyed a general directed influence sequence with: i) dynamic social variables and ecological state variables as mostly influential on the overall system, ii) provisioning and cultural services as target variables, and iii) regulating services and biodiversity parameters as mostly impacted variables. We demonstrated that the INF holds potential to deliver synthetic assessments of ES relations through spheres (ecological / social), scales (local to global) and opinions (depending on stakeholder groups). We stress its potential as a tool for increased understanding and supporting communication on complex social-ecological systems as well as for supporting environmental management.

1 Introduction

The ecosystem service (ES) concept has been acknowledged as relevant for bridging the still existing gap between ecological management and social development (Chan et al. 2012, Martín-López et al. 2014). In particular, working on ES trade-offs and synergies (respectively, consistent negative and positive co-variations (Mouchet et al. 2014)) could support more sustainable management of environmental resources, required both for maintaining desired ecosystem functioning and enhancing human well-being (Rodríguez et al. 2006, Kareiva et al. 2007, Luck et al. 2012).

There is a growing agreement that the pivotal function of ES arises from their interface position within the social-ecological system (MEA 2005), as they account jointly for biophysical and socio-cultural factors (Bennett et al. 2009, Reyers et al. 2013) and associated value-domains (Martin-Lopez et al. 2014). This ability is described specifically by a combination of three facets (Burkhard et al. 2012, Villamagna et al. 2013, Bagstad et al. 2014), that, in current lack of consensus on precise terminology, will be hereafter referred to as ES potential supply, demand and actual supply facets. First, potential supply is defined as the ecosystem potential “capacity to supply services” (Bastian et al. 2012), due to the combination of geophysical and ecological characteristics in the current land cover matrix. It has been also referred to as “capacity” (Villamagna et al. 2013, Schröter et al. 2014) or “managed supply” (Geijzendorffer et al. under review). Second, demand is understood as the “social demand for using a particular ES in a specific area” (García-Nieto et al. 2013) and represents “the amount of service desired by society” (Villamagna et al. 2013). Third, actual supply depicts the actual encounter of demand and potential supply; it has also been called “budget” (Burkhard et al. 2012), “flow” (Villamagna et al. 2013, Schröter et al. 2014) or “match” (Geijzendorffer et al. under review). Alternative terminology for all three facets can be found in the interesting reviews by Villamagna et al. (2013) and Geijzendorffer et al. (under review). Those three facets apply for all ES notwithstanding their category (provisioning, cultural, regulating).

Many authors have addressed ES trade-offs and synergies from the perspective of their potential supply (e.g. Anderson et al. 2009, Raudsepp-Hearne et al. 2010, Bai et al. 2011), to provide the better ecological understanding required for robust management decisions (Kremen 2005). Furthermore, acknowledging the necessity of taking into account social components, some have integrated demand into trade-off assessments for a single ES (e.g. pollination (Schulp et al. 2014)) or for multiple ES (Palomo et al. 2013, Hauck et al. 2013, García-Nieto et al. 2013). Finally, the actual ES supply has also been considered to characterise the (mis)matches between supply and demand (recently Bagstad et al. 2014, Van der Biest et al. 2014).

Several conceptualisations of trade-offs and synergies have been proposed. Among these, Bennett et al. (2009) proposed a framework distinguishing direct relations between ES from indirect relations linked to external factors. Rives et al. (2012) adapted this framework by explicitly distinguishing interactions arising from the ecosystem from those linked the social system to analyse forest policy reforms in Niger. As a complementary approach, Kandziora et al. (2013) proposed to describe main supporting, reducing and feedback links between pairs of ES using direct interrelation matrices, and illustrated their interests for typical central European landscapes. However, while ES facets have been considered among the many criteria proposed to characterise and classify trade-offs and synergies between ES (Mouchet et al. 2014, Van der Biest et al. 2014), most trade-off and synergy assessments have been carried out irrespective of the distinction between potential supply, demand and actual supply ES facets.

To go a step further, a more detailed framework is therefore needed that describes appropriately influence relationships among ES and external variables, both social (e.g. land allocation) and ecological (e.g. specific biophysical conditions). In this study, our main objective was to expand the ES trade-off framework (Bennett et al. 2009) in order to explicitly consider ES associations within and between potential supply, demand and actual supply facets, leading to what we called the “Influence Network Framework” (INF). To test the operational implementation of this INF and reveal interactions perceived as most influential in environmental management, we used a consultative process in the French Alps. Research questions guiding this process are summarized in Figure 1. Interactions were depicted as networks considering influences both within and among the three ES facets. Based on these, the propensity of each category of variables (namely ES categories, biodiversity, social and ecological variables) to influence the overall system or to be impacted by it was quantified. We calculated the ratio of emitted on received influences, and synthetized the results as a general sequence of influence. Overall, we demonstrate the value of the simple decomposition of relationships and of the consideration of ES facets for improving understanding by disentangling complexity. Lastly, we discuss the interests and potentialities of the framework, illustrated by insights from the French Alps assessment.

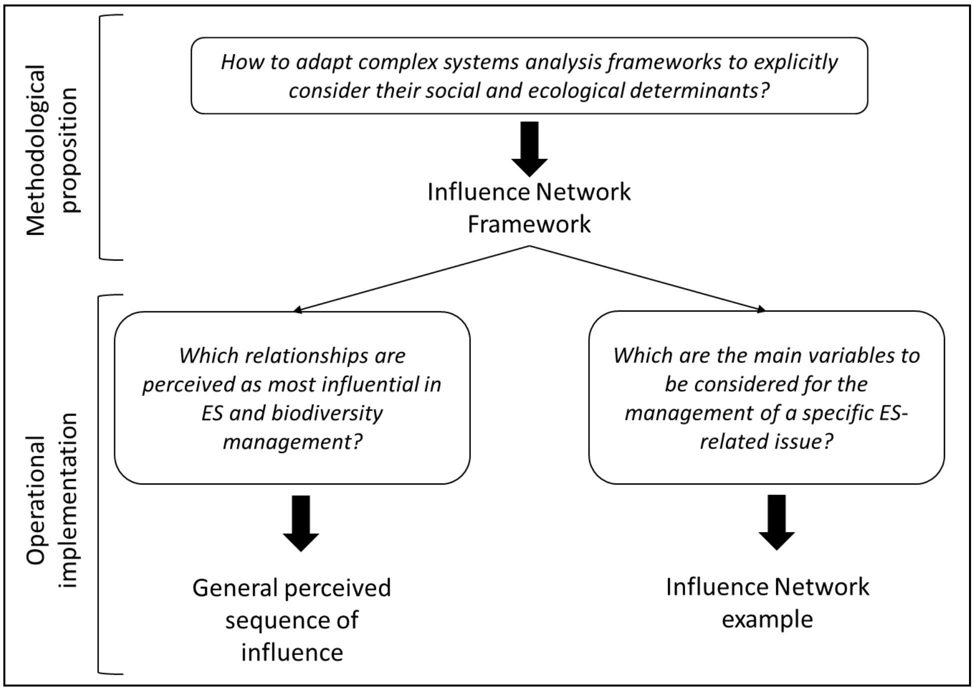

Figure 1: Research questions explored and illustrated by the results of a participative process in the French Alps. ES: ecosystem services

2 Methods

2.1 Study area

Our analysis focused on the French Alps (Figure 2), which encompass 52 149 km² over nine « départements », the core administrative level in France. The French Alps are the western part of the Alpine arc and their complex topography encompasses elevations below 100 m to Mont Blanc culminating at 4810 m. Dominant land cover types are forests and semi-natural areas (67%), followed by arable lands (27%) mainly in the western broad valleys and piedmonts, concentrating artificial covers over a restricted area (5%) (following Corine Land Cover 2006 categories). High-density urban areas in the valleys, where labour market is concentrated, contrast with more isolated or more rural areas. The broad latitudinal climate and vegetation gradient has had historical consequences on social dynamics and economic activities. Due to natural constraints (altitude, climate, slope inclination), the eastern part of the French Alps has been dedicated to livestock farming favouring cultural landscapes. In the South and in the longitudinal valleys of the western Alps, more gentle natural conditions permit mixed or field cropping. Within this regional matrix, the steepest and most constrained areas (e.g. highly erodible soils) have seen continuous depopulation since World War II resulting in a sharp decline in farming activities, and the subsequent closing of landscapes by natural afforestation. Forms of tourism are also contrasted. In the Northern Alps, tourism intensity is high, mainly during winter time, thus impacting high altitude sensitive areas through infrastructure development. In the Southern Alps, tourism is usually more rural and small-scale. Altogether, the diversity of biophysical and human uses is responsible for the high variety of biodiversity, ecosystems and ES across the entire area (Tappeiner et al. 2008, Crouzat et al. in review).

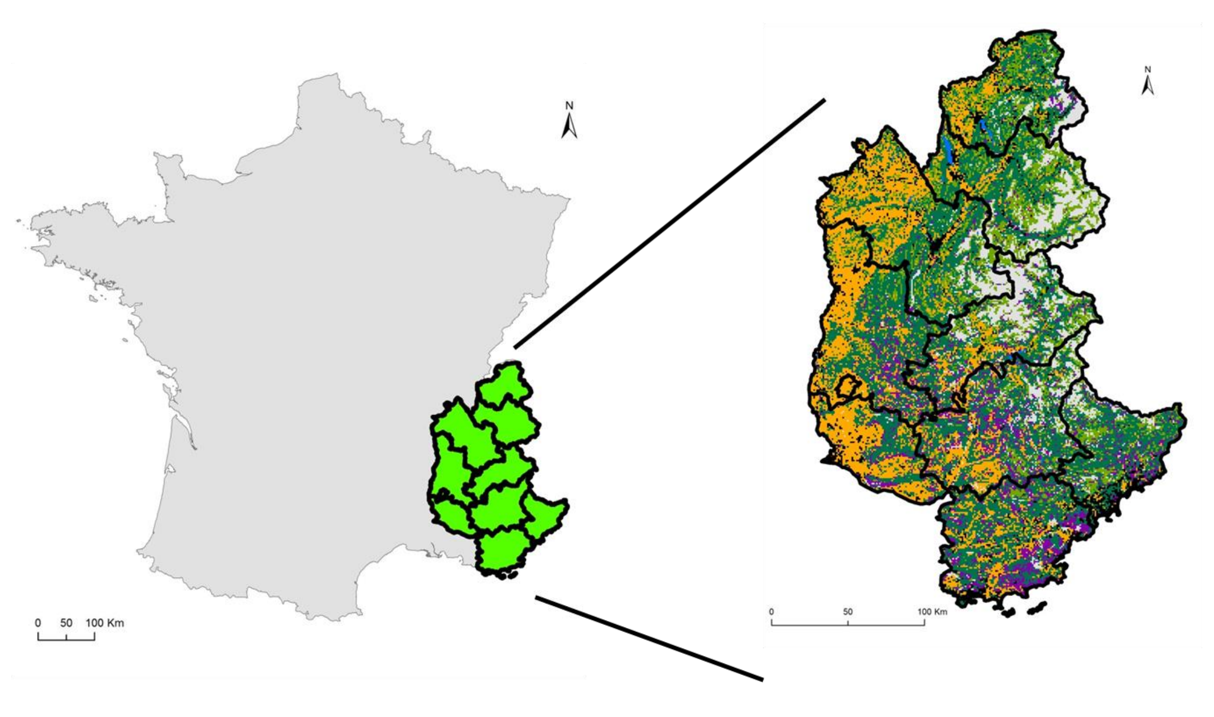

Figure 2: The French Alps in France (left) – Main land covers in the French Alps (right): black = artificial areas, orange = agricultural areas, light green = grasslands and pastures, dark green = forests, purple = semi-natural areas, grey = open spaces with scant vegetation, blue = wetlands and waterbodies. Dark delineations represent administrative boundaries of "départements".

2.2 The Influence Network Framework (INF)

Bennett et al. (2009) proposed a framework to distinguish between "true" direct interactions between pairs of ES and indirect relations arising from external drivers, in order to better understand the mechanisms underpinning trade-offs and synergies. This framework described six configurations resulting from combinations of the strength of ES interaction (weak – medium – strong) and the impact of external drivers on ES (independent – shared). Complexity of interactions increased along the various configurations (1 to 6). Rives et al. (2012) further showed that this framework can be adapted to characterise influence relationships between ES by specifying the nature of interactions (competition or mutual benefit) and their origin (social system or ecological system).

To go one step further in the development of this original framework, we suggested that more comprehensive understanding of the social-ecological system would be gained by formally describing interactions specific to the three ES facets (Figure 3). In this Influence Network Framework (INF), ES interactions were characterised as unilateral influences when one ES influenced a second one without major feedback, or as mutual influences when both ES influenced each other, both within and between ES facets. External variables and biodiversity were considered as independent influencing variables when they impacted a single ES and as shared influencing variables when they impacted pairs of ES. In turn, biodiversity and external variables could be impacted by ES.

Positive influences represented the case when one ES would foster the potential supply, demand or actual supply of a second ES or when the external variable would benefit to the ES. Negative influences were used to represent the opposite trends. Varying influences were needed to express influences that had both positive and negative aspects, and also to describe influences that could vary depending on magnitude of change, intensity of practises, etc.

External variables were defined as social variables if they were related to human choices (e.g. land allocation choices, policy measures, specific practises in agriculture and forestry, property rights or evolutions in social demand). They were complemented by ecological variables describing biophysical features (e.g. temperature, precipitation, soil type or slope). These biophysical variables can be considered mainly as stable in the perspective of this assessment. In addition, the 'biodiversity' variable was singled out to account for the role of particular species (e.g. burrowing animals damaging agricultural production, soil biodiversity responsible for its fertility). Biodiversity was also considered as a whole to describe for example general impacts of urbanisation or the importance of biodiversity for landscape aesthetics.

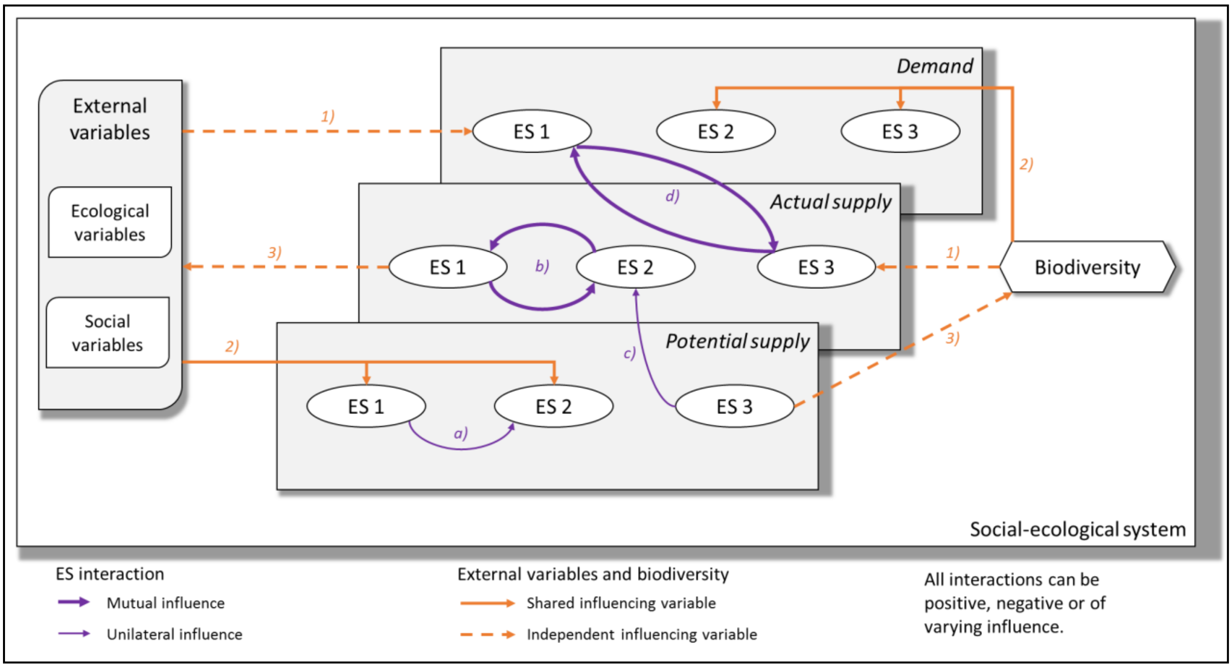

Figure 3: Influence network framework (INF). The INF describes influence relations between ES, biodiversity and external ecological and social variables. ES are described explicitly by their three facets: potential supply, demand and actual supply. Within each facet, ES interactions are unilateral when one ES influences a second one without major feedback (a) and mutual when both ES influence each other (b). ES interactions also concern distinct facets, both with unilateral (c) and mutual (d) influences. External variables and biodiversity are independent influencing variables when they impact a single ES (1) and shared influencing variables when they impact pairs of ES (2). In turn, biodiversity and external variables can be influenced by ES (3). All relations can be positive, negative or of varying influence.

2.3 Data sources and analysis

Our approach was grounded in a consultative process that used the INF as a descriptive and analytic tool. Based on qualitative data obtained from regional experts (Figure 4), we explored how ES were perceived to relate to each other and to external variables in the specific area of the French Alps.

In our methodological design (Figure 5), the consultative phase comprised three steps. In the first step, eight semi-structured interviews were used to draw up a comprehensive overview of how ecosystems were conserved, used or impacted. Specifically, we assessed demands for ES and biodiversity and explored main determinants of their actual supply. As a second step, fifteen attendants debated in a focus group the synthesis of first step results. Discussions on positive and negative consequences of actual human uses on biodiversity and ES potential supply were conducted successively focusing on specific landscapes: forested areas, agricultural landscapes, open (semi-)natural spaces and artificial areas. The third step used twelve semi-structured individual interviews to further investigate ES influence networks.

From a list containing ES discussed in the two previous steps, each interviewee selected and justified up to ten ‘highly important’ ES, before detailing main variables influencing and being impacted by those ES.

Stakeholder sampling was intentional and reflected the need for “information-rich cases” (Coyne 1997, Menzel and Buchecker 2013): we focused on experts representing different domains of competence required in this analysis, following a classical case-study research approach (Eisenhardt 1989). In the third step, we estimated that information gathering was sufficient after twelve interviews as we reached saturation of information (Eisenhardt 1989, Lugnot and Martin 2013). Semi-structured interviews were chosen in the first and third steps as they are known to provide “reliable, comparable qualitative data to get a practicable understanding of stakeholders’ knowledge, intentions and actions” (Lugnot and Martin 2013).

The fourth step of our methodological design consisted in post hoc treatments and data analysis. All interviews and discussions were recorded, transcribed and coded following a deductive qualitative content analysis process (Elo and Kyngäs 2008, Lugnot and Martin 2013). Simple relationships linking two ES, or one ES and an external variable, were formalised by considering jointly outputs from the three consultative process steps. Influences were specifically attributed to ES facets. As a comprehensive post hoc treatment of stakeholder perceptions, we calculated the ratio between the number of distinct emitted influences and the number of distinct received influences by categories of variables (namely ES categories, biodiversity, social and ecological variables). By distinct we mean without taking into account the number of stakeholders having mentioned each influence. The higher the ratio, the more the variable influenced the system and the lower the ratio, the more the variable was impacted by the system. Finally, we designed influence networks regrouping all factors sharing a direct link with either of the facets of focus ES, thus operationalising the INF.

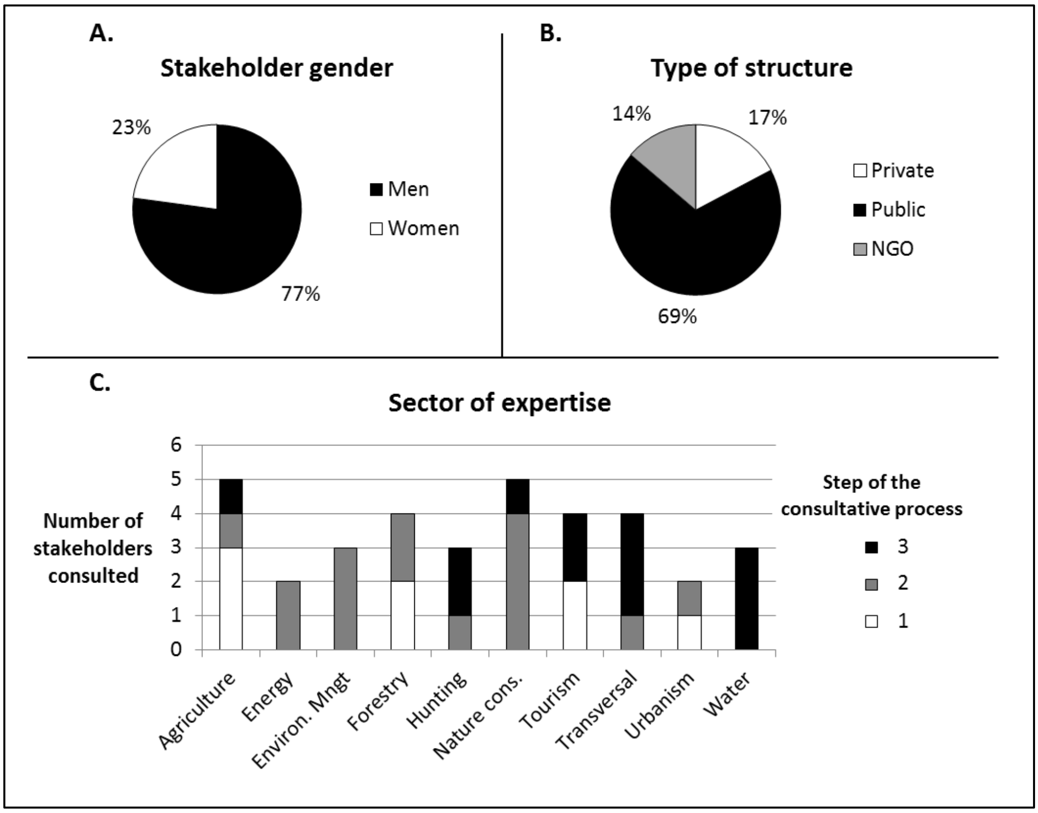

Figure 4: Profiles of stakeholders consulted in the operational implementation of the Influence Network Framework: gender (A.), type of structure (B.) and main sector of expertise (C.). Abbreviations: Envirn. Mngt stands for Environmental Management, Nature cons. stands for Nature conservation.

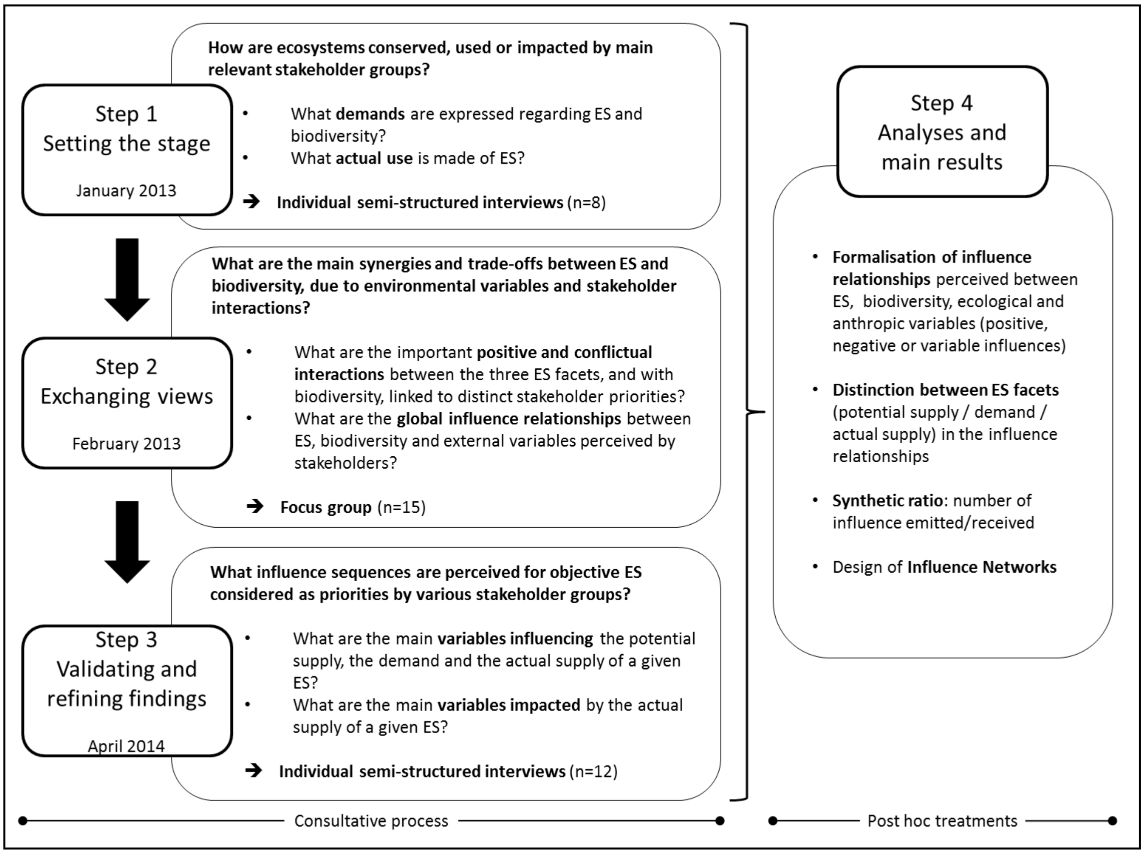

Figure 5: Consultative process steps and related questions to explore ecosystem service (ES) networks in the French Alps using the Influence Network Framework (INF)

3 Results

3.1 Exploring the three facets of ES

Stakeholders chose to discuss influences relationships concerning 5 provisioning services, 5 cultural services and 10 regulating services (Table 1). External variables describing other components of the social-ecological system were classified as social variables or ecological variables (Table 2).

Table 1: ES discussed by stakeholders during the consultative process and specification of their three facets. Provisioning ES appear with a pink background, cultural ES with a green background and regulating ES with an orange background.

ES	Potential supply	Demand for	Actual supply
Fresh water supply	Freshwater available	Water needed for irrigation, industry, domestic consumption…	Volume of water from the ecosystem actually used
Wood energy	Logging residues from wood harvesting	Accessible and profitable logging residues as renewable energy source	Amount of wood actually harvested in forests to be used for biomass energy production
Hydro energy	Medium to large water bodies in steep areas	Local, "green", profitable and renewable energy	Energy produced from hydroelectric plants
Wood production	Biophysical potential to grow harvestable timber	Accessible and profitable timber	Amount of wood actually harvested in forests
Agricultural productions	Biophysical potential to grow harvestable agricultural products	Specific agricultural products	Crop and fodder yields
Leisure hunting	Presence of wild game species	Accessible, undisturbed and numerous game	Game actually killed
Iconic species	Abundance and richness of specific wild species	Social interest for designating iconic species	Actual designation of iconic species
Landscape aesthetics	Potential landscape aesthetic quality	Satisfaction obtained from contemplating particular landscapes	Landscapes with aesthetic quality that actually fulfil the social need of aesthetic enjoyment
Nature tourism	Attractive (semi-)natural areas	Accessible, secured and varied outdoor activities	Actual number of people enjoying outdoor tourism
Educative value	Large gradient of biophysical conditions and human activities from which environmental education arise	Awareness and knowledge of ecosystems functioning	Actual number of people with increased environmental awareness
Biological control of pests	Presence of predator species	Agricultural sector demand for pest control	Actual control of agricultural pests by natural predators in relevant areas
Soil erosion mitigation	Soil retention and protection by plant cover, notwithstanding human value and uses of the area	Demand for in-situ soil conservation, unsilted water and absence of mudslides	Amount of soil erosion actually prevented by plant cover in managed and human-occupied areas
Gravitational hazards mitigation	Presence of natural protective elements from plant cover (forests - pastures) in areas exposed to gravitational risk but notwithstanding its human value and uses	Protection of human activities and infrastructures	Actual protection (or damage limitation) of human infrastructures from gravitational hazards by natural elements
Fire risk mitigation	Specific vegetation and land configuration reducing fire spread, notwithstanding human value and uses on the area	Protection of human activities and infrastructures	Actual protection (or damage limitation) of human infrastructures from fire hazards
Maintenance of soil fertility	Stock and recycling of nutrients needed for biomass growth, depending on above-ground biomass, soil biodiversity and edaphic conditions	Ability of soils to provide nutrients to grow biomass as required by human land use choices	Actual adequacy between natural soil functioning (i.e. without inputs) and human requirements
Maintenance of water quality	Ecosystem ability to retain pollutants and nutrients from water fluxes, depending on plant cover and edaphic conditions	Fresh water corresponding to quality standards set by legislation	Amount of pollutants and nutrients actually retained and not reaching water bodies

Pollination	Floral resources and habitats for wild pollinators	Required pollination of agricultural areas (crops, orchards…) by wild pollinators	Amount of crops and cultures actually pollinated by wild pollinators
Flood risk mitigation	Ecosystem ability to buffer river discharge after heavy precipitation events, depending on plant cover and edaphic conditions	Protection of human activities and infrastructures from flood risks	Actual protection (or damage limitation) of human infrastructures from flood risks by natural elements
Water quantities regulation	Ecosystem ability to regulate the runoff regime in a river catchment, depending on plant cover and edaphic conditions	Limited runoff, stable water stock in soils and stable water flows	Actual regulation of water flows and stocks in soils
Global climate regulation	Ability to store and sequester carbon in ecosystems, depending on above and below ground biomass, dead organic matter stocks and soils	Limited global amount of greenhouse gases in the atmosphere	Amount of carbon stored and sequestered by ecosystems

Table 2: Social and ecological variables considered by interviewees to describe influence relationships with ES and biodiversity in the alpine social-ecological system.

Social variables	Ecological variables
Policy (including protective status)	Biophysical conditions of mountain areas (slope – altitude – climate – seasonality – vegetation types …)
Urbanisation	Landscape diversity: Heterogeneous and open landscapes
Society evolution (e.g. age – balance between rural / urban population – evolution in social demand…)	Anthropogenic-induced changes in precipitation, temperatures etc.
Economic profitability and structuring of the activity sector	
Diversity and management of human uses depending on the provisioning capacity of ecosystems (agriculture / forestry…)	

3.2 Testing the Influence Network Framework (INF) operational potential

Picking from the 200 simple influence relationships extracted from the consultative process (results not shown), we exemplified relations in the INF within each of the three ES facets (Figure 3, relations a, b, 1, 2): potential supply (Figure 6), demand (Figure 7), actual supply (Figure 8). We also exemplified relationships between facets (Figure 3, relations c, d, 1, 2, Figure 9). Supporting Information S1 to S4 provide respectively further descriptions of each of these influence relationships (at the end of the manuscript in the Appendices from Chapter II (Section B)).

ES interactions both within and between facets presented mutual influences that could reinforce each other (i.e. two synergies or two trade-offs). For instance, supply of biological control of pests was perceived to increase agricultural yields, which in turn provided more habitats and resources for natural predators (Figure 6.5). Regarding negative influences, demands for wood production and leisure hunting were mentioned as conflicting as they relied on low vs. high wild ungulate abundances (Figure 7.5). In addition, ES mutual influences could have antagonist effects, i.e. one synergy and one trade-off. For example, increased maintenance of water quality enabled more actual fresh water supply at reduced costs, while more water extraction could lead to scarcity and thus to a diminished water quality, according to stakeholders consulted (Figure 8.6). Similar patterns were observed for the influence of external shared influencing variables, which could affect ES in the same way or in opposite trends. Indeed, urbanisation was mentioned as negative both for the presence of iconic species and for the maintenance of water quality (Figure 6.2), while mountain biophysical conditions were described as a positive factor of specificity for the demand of nature tourism and as a negative factor for potential supply of agricultural production due to limiting biophysical constraints (Figure 9.2).

Influencing variables perceived as important for all three facets could be either ecological (e.g. Figure 7.3: high summer temperatures, affecting positively the demand for summer nature tourism due to cooler temperatures at altitude) or social (e.g. Figure 6.6: deep ploughing in agricultural practises, that was mentioned as negative both for soil fertility and erosion mitigation potential supply).

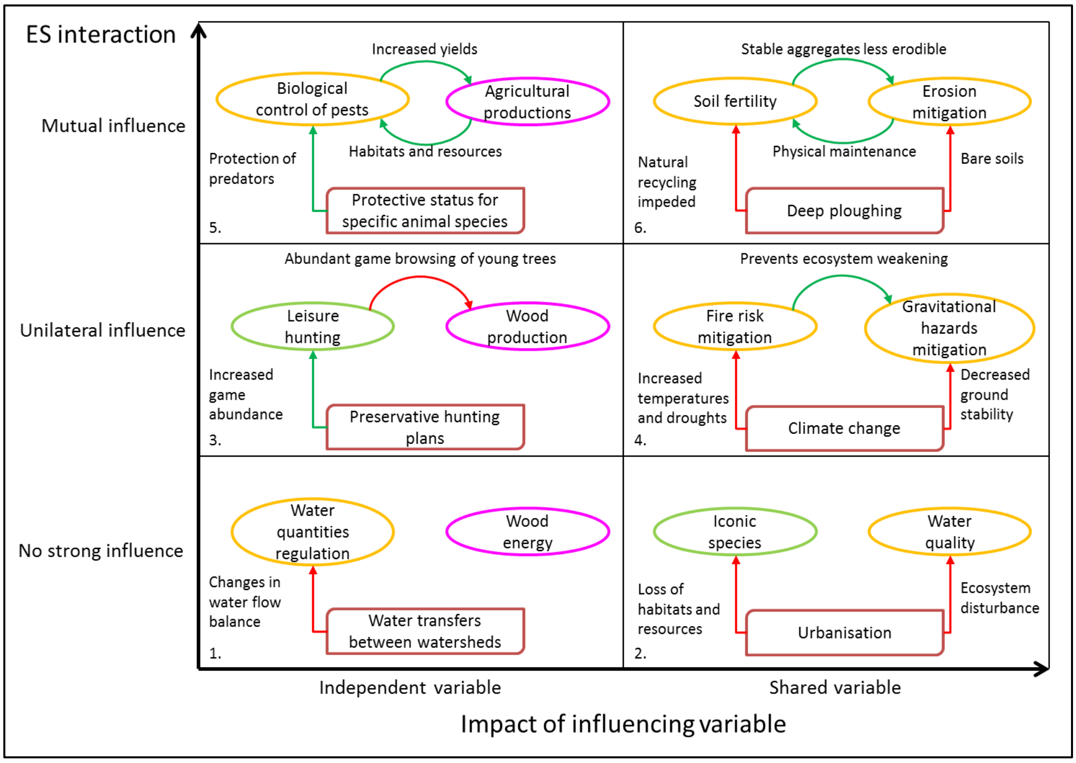

Figure 6: Influence relationships between ES potential supply facets exemplified from a consultative process results. Provisioning ES are circled in pink, cultural ES in green and regulating ES in orange. Green arrows represent a positive influence, red arrows a negative influence, and orange arrows describe influences with either positive and negative aspects, or varying ones. Bottom rectangles represent external influencing variables.

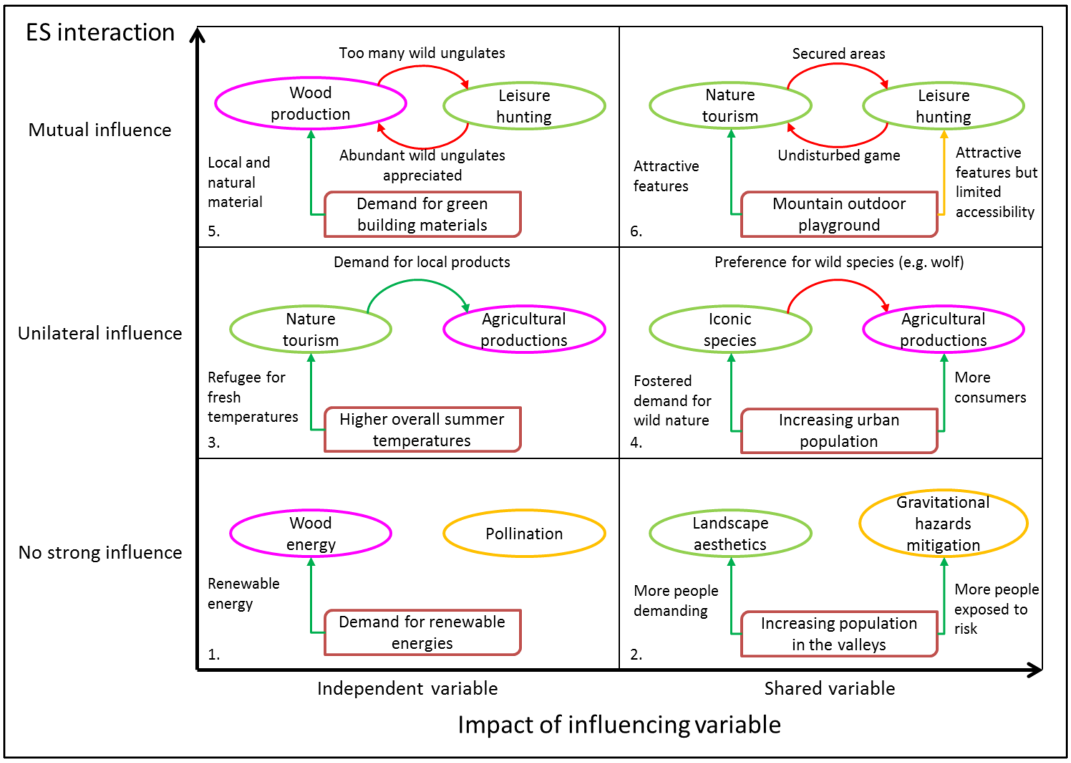

Figure 7: Influence relationships between ES demand facets exemplified from a consultative process results. Provisioning ES are circled in pink, cultural ES in green and regulating ES in orange. Green arrows represent a positive influence, red arrows a negative influence, and orange arrows describe influences with either positive and negative aspects, or varying ones. Bottom rectangles represent external influencing variables.

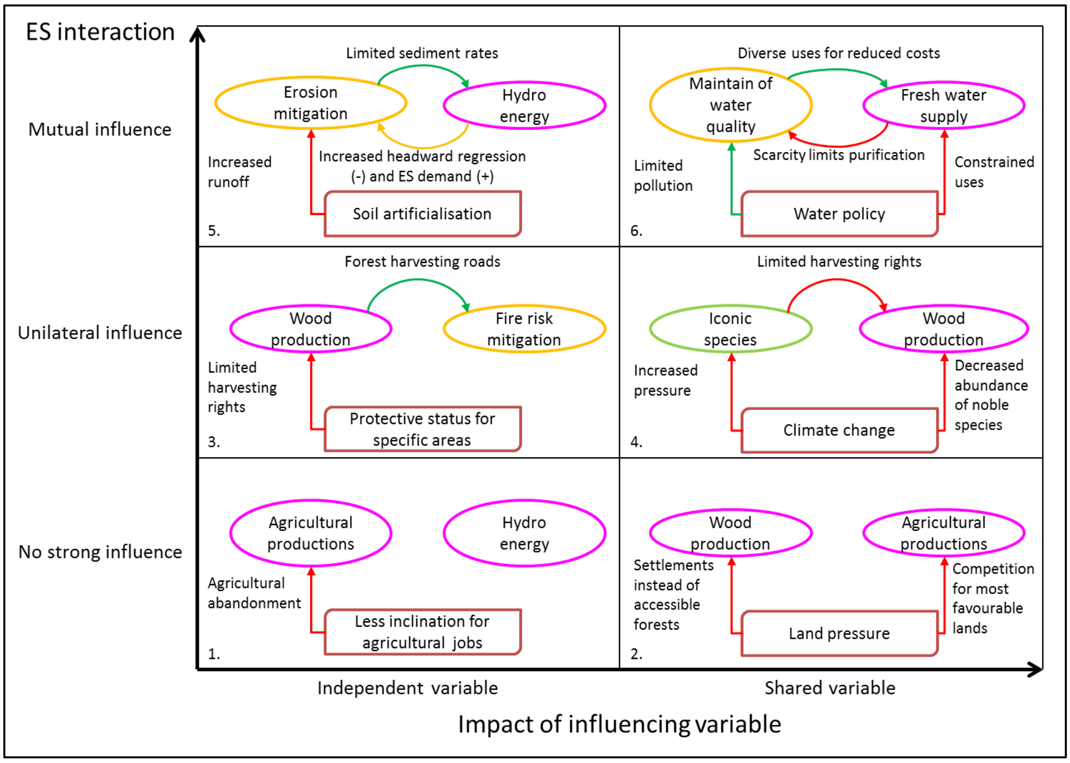

Figure 8: Influence relationships between ES actual supply facets exemplified from a consultative process results. Provisioning ES are circled in pink, cultural ES in green and regulating ES in orange. Green arrows represent a positive influence, red arrows a negative influence, and orange arrows describe influences with either positive and negative aspects, or varying ones. Bottom rectangles represent external influencing variables.

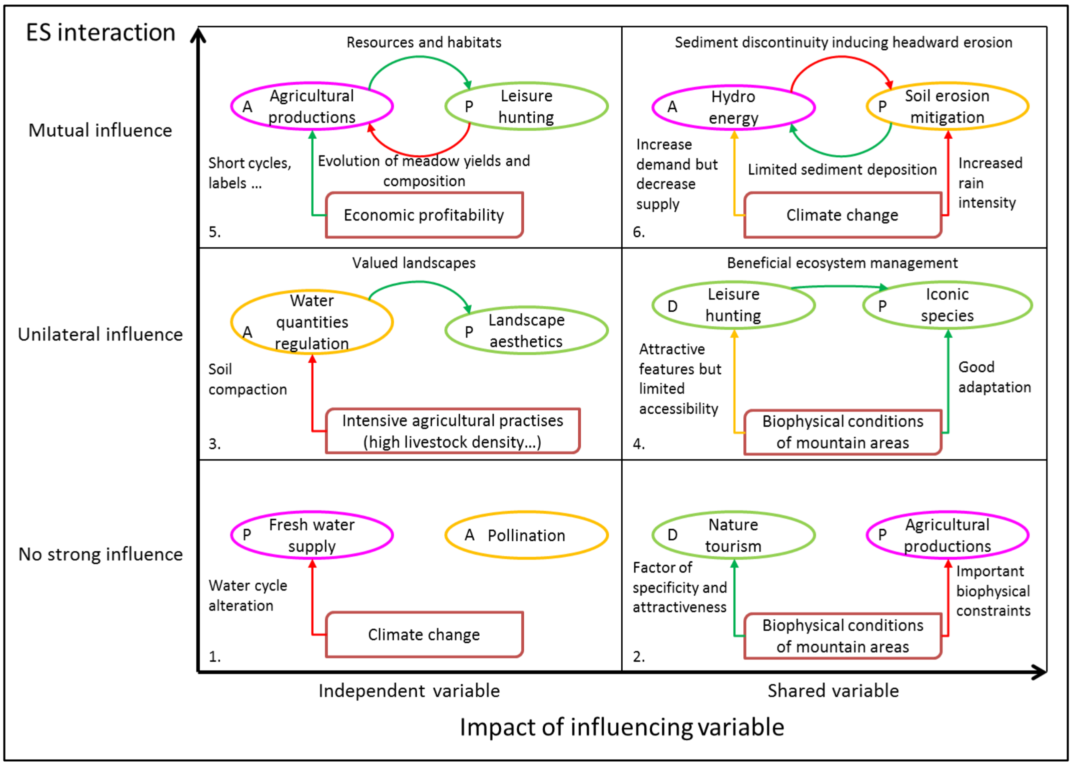

Figure 9: Influence relationships between ES facets exemplified from a consultative process results. P stands for potential supply, D for demand and A for actual supply facet. Provisioning ES are circled in pink, cultural ES in green and regulating ES in orange. Green arrows represent a positive influence, red arrows a negative influence, and orange arrows describe influences with either positive and negative aspects, or varying ones. Bottom rectangles represent external influencing variables.

3.3 Example of INF focused on leisure hunting

By aggregating simple influence relationships, we were able to design influence networks showing in an explicit manner the many parameters and mechanisms related to trade-offs and synergies between ES and biodiversity. Figure 10 proposes one such network focused on leisure hunting to illustrate the interests of the INF.

The leisure hunting influence network showed shared influences with all ES categories (regulating, provisioning and cultural) as well as with ecological and social variables. Some influences concerned similar facets of ES, while other relationships connected different facets (e.g. actual leisure hunting and supply of biological control of pests).

The INF highlighted opportunities for stakeholder synergies. As an example, the actual supply of resources and habitats for game species by agricultural areas could prompt farmers to adopt wildlife friendly practises to enhance game abundance (i.e. leisure hunting potential supply). This opportunity has actually been formalised through specific farmer voluntary engagement, based on incentives from the hunters' federation ('Agrifaune' program). In addition, the INF exposed reasons for conflicts between stakeholders. Indeed, the conflict mentioned between hunters and nature tourists arose from antagonist demands, with hunters requiring game undisturbed by tourists while these complained from insecurity during hunting periods. Managing this situation would be a social process, requiring stakeholder conciliation and more formal rules to frame their practises. Those examples illustrate how differentiating between ES facets allowed us to precisely identify the origins of ES synergies and trade-offs, a required step for promoting or limiting them. This has been considered essential to identify "ecological leverage points where small management investments can yield substantial benefits" (Bennett et al. 2009).

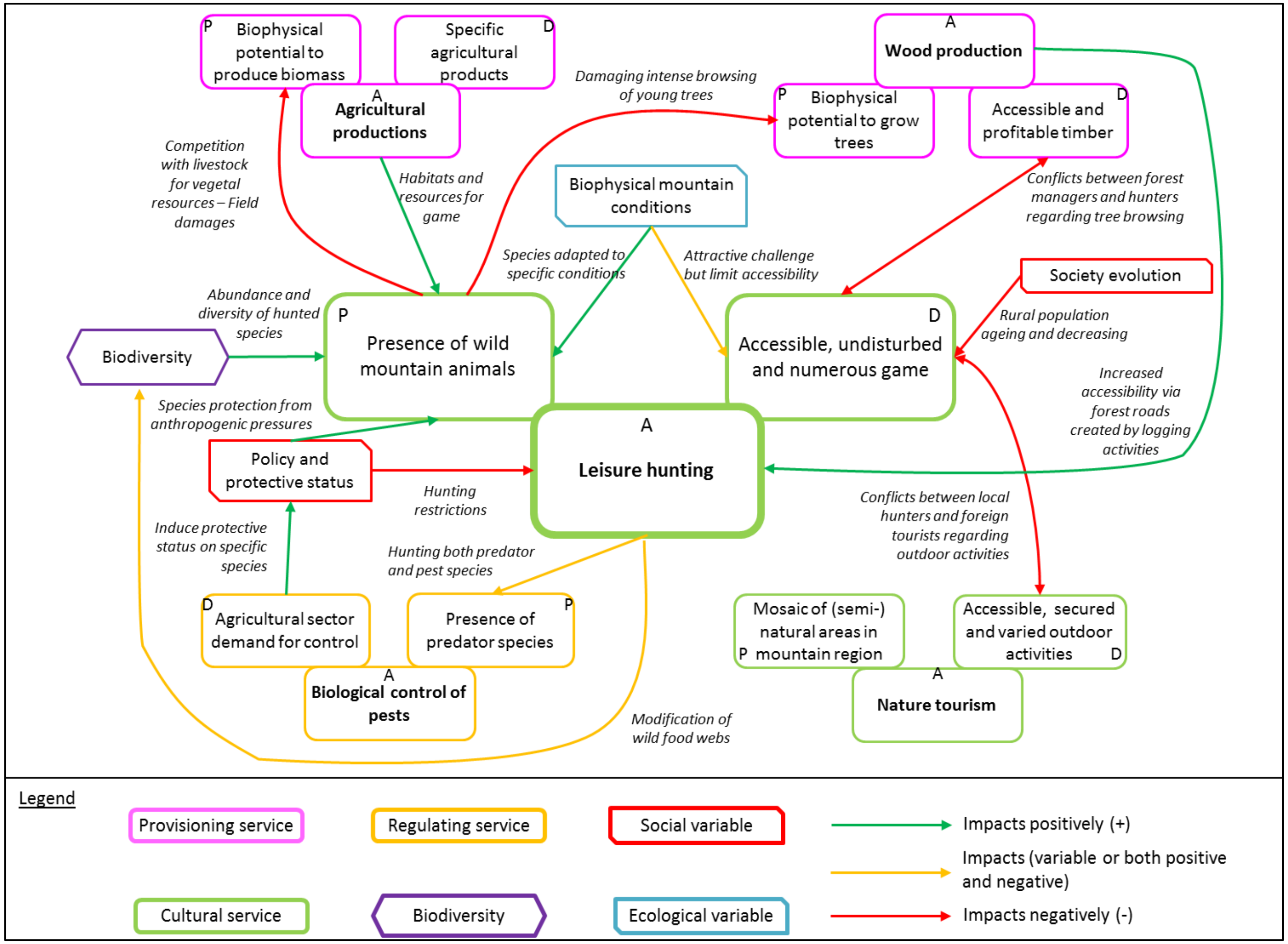

Figure 10: Leisure hunting influence network as constituted by some examples of relations described by the consultative process over the French Alps. Only direct relations from or to leisure hunting service are exposed, and all mentioned relations were not included to limit complexity. ES facets are described by "P" for potential supply, "D" for demand and "A" for actual supply. Provisioning ES are circled in pink, cultural ES in green and regulating ES in orange. Biodiversity is represented by purple hexagons. Green arrows represent a positive influence, red arrows a negative influence, and orange arrows describe influences with either positive and negative aspects, or varying ones. Red and light blue rectangles represent external factors of influence, respectively social variables and ecological variables.

3.4 Overall influence ratio

As a further post hoc treatment, the ratio of emitted influences on received influences showed distinct features for external variables, different categories of ES and biodiversity (Figure 11). Social and ecological external variables had a ratio greater than 1.5, expressing that stakeholders perceived them as most influential on the system while largely unaffected by other variables. However, the reasons why they were considered as unaffected varied, as ecological variables were described as quasi fixed due to external biophysical constraints (soils, slopes…) while social variables only reflected the current socio-cultural setting and had the ability to evolve. Both cultural and provisioning services had ratios comprised between 0.5 and 1.5, meaning that they both received and emitted a fairly equivalent amount of influences. Finally, biodiversity and regulating services presented the lowest ratio, smaller than 0.5, showing that stakeholders perceived them as under influence of the whole system but of limited importance for the influence they could exert on other variables. Thus, in the general influence sequence, we classified social and ecological variables as mostly influencing variables, cultural and provisioning services as target ES and biodiversity and regulating services as impacted variables.

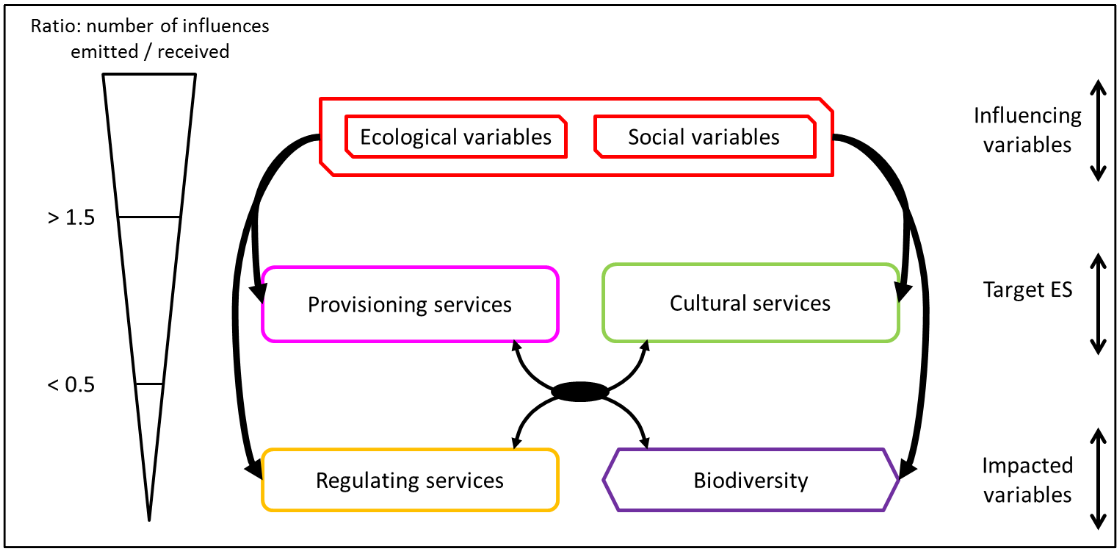

Figure 11: Overall influence sequence summarizing perceived influence relationships as described by the participative process in the French Alps.

4 Discussion

We demonstrated that the INF was suitable for qualitatively describing trade-offs and synergies concerning ES, respective to their distinct facets, and other components of the social-ecological system, namely biodiversity, social and ecological variables. This framework was applicable for both simple influence relationships between pairs of variables and for more complex influence networks including multiple components. It provided a comprehensive view of how social and ecological systems interacted and offered a basis to place stakeholder interactions in a broader context. Furthermore, the INF allowed us to synthesize as an overall sequence of influence how stakeholders perceived the links between ecological and social systems. We now discuss main insights at conceptual and operational levels, considering four issues: i) the origins and consequences of discrepancies between actual and perceived ecological influences, ii) the interests of integrating multiple stakeholder perceptions, iii) the advantages of making explicit distinction between the three ES facets, and iv) challenges and opportunities of addressing complexity.

4.1 Discrepancies between perceived and actual ecological influences

The overall sequence of influence which came out from French Alps participative process (Figure 11) showed that regulating services and biodiversity were generally described as undergoing many influences from the system while exerting a low influence on other components. This result is consistent with other analyses of stakeholder perceptions. For instance in a case study focused on the region of Krummhörn, Germany, a lack of awareness regarding ecosystems ability to mitigate natural hazards was observed (Karrasch et al. 2014); likewise biodiversity was found to be undervalued by local residents and tourists in a Mediterranean semiarid region (Almeria province, Spain) (Castro et al. 2011). Thus, influences perceived by stakeholders may differ from actual ones, as regulating services are known to be necessary for other ES to be supplied (Villamagna et al. 2013). For instance, while agricultural production was perceived as impacting both the potential and actual supply facets of pollination service by wild pollinators, the opposite relationship (positive influence from pollination to agricultural service) was not mentioned as important, although the absence of insect pollination would decrease total European crop production by ~30% (Zulian et al. 2013).

Four hypotheses could explain this lack of consideration.

First, stakeholders could perceive regulating services as taken for granted, overall in areas of high environmental quality as the French Alps (EEA 2002, Crouzat et al. in review) where ecosystem ability to supply ES, and mostly provisioning and cultural ES, may not have been degraded (yet) to perceived threatening levels (Villamagna et al. 2013).

Second, many authors observed a higher difficulty for stakeholders to grasp the importance of regulating services and biodiversity (Lewan and Söderqvist 2002, Villamagna et al. 2013): they are considered out of their sphere of experience and are more difficult to perceive by the senses (Iniesta-Arandia et al. 2014). Indeed, they are often intermediate services contributing to the supply of other ES and not final ES from which stakeholders directly benefit (Boyd & Banzhaf 2007, Fisher et al. 2009). The same reasoning could apply for biodiversity features.

Our third hypothesis considers that some stakeholders trust technological solutions to compensate for negative budgets between actual ES supply and society demand (Schneiders et al. 2011). For example, protective dikes can mitigate floods, commercial beekeepers can be mobilised where wild pollinators are insufficient and fertilizers can be used to stimulate depleted soils. However, such technological responses are sufficient only in the short term and for small depletion rates. Regulating services are essential for ensuring ecosystem resilience and avoiding dramatic shifts in ES supply (Bennett et al. 2009, Hauck et al. 2013).

Fourth, some authors advocated that use of the ES concept would be in essence focused on influences from the social system onto the ecosystem, thereby necessarily focusing our influence sequence on “how human actions and resources needs affect the ecological system” (Binder et al. 2013). However, alternative visions of the concept have been proposed, describing ES as rising from a ‘cascade’ rooted in biophysical structures and processes (Haynes-Young and Potschin 2010) or insisting on the importance of the ecological risks and returns associated with ES supply (Abson and Termansen 2011). We contend that using an ES-based framework does not necessarily blind to complexity (Norgaard 2010) as multiple facets and external variables can be jointly considered (Briner et al. 2013).

4.2 Uncovering multiple perceptions of the social-ecological system

Here we synthesized perceptions by the diverse groups of stakeholders (Figure 4) into a single sequence of influence, i.e. notwithstanding the different points of view that had been expressed. A more comprehensive view of the system could be obtained by explicit consideration of multiple stakeholder profiles (Lamarque et al. 2011). This is consistent with other studies (e.g. Castro et al. 2011, Lugnot and Martin 2013; Iniesta-Arandia et al. in press) where different stakeholder groups presented various priorities in environmental management and demonstrated varying perceptions and knowledge about social-ecological system dynamics. In particular, regulating services were highly prioritized by stakeholders in rural systems to maintain other ES (Martín-López et al. 2012; Hauck et al. 2013; Iniesta-Arandia et al. 2014) as well as their personal wellbeing (Oteros-Rozas et al. 2013; Zagarola et al. 2014). Moreover, exposing the differing relationships perceived represents an alternative entry point on territorial conflicts that could be used as a tool for collective learning and management (Lamarque et al. 2014, Felipe-Lucia et al. submitted). Subsequently building a common understanding of the social-ecological system facilitated collective management processes. Hence, there is a future challenge to apply the INF methodological tool to account for multiple stakeholder profiles and related different associations between ES.

4.3 Advantages of multi-faceted ES analysis

Going a step further than working on widely-adopted ES categories (provisioning, cultural, regulating), the inclusion of ES facets in the INF holds at least four advantages.

First, our analysis demonstrated that distinguishing between ES facets is necessary to embrace the complexity of ES relationships. As one example, consider relationships from nature tourism onto wood production. Actual nature tourism was described as negative to wood production potential supply, as increasing off-piste skiing damages young trees and thereby limits wood production. This conflict could be addressed by a conciliation process gathering representatives from the two sectors and further by ensuring applicability of restriction access if needed. In parallel, demand for nature tourism also negatively impacts actual wood production, as some alpine municipalities limit logging due to tourist demand for forests without explicit, and negatively perceived, signs of logging. As an answer, helicopter harvesting in highly touristic areas near Mont-Blanc have been adopted. As adequate management measures to problems differ, addressing trade-offs should be eased by in-depth understanding of their determinants, explicitly exposed with ES facets. Moreover, interestingly, formal disaggregation between ES facets from stakeholders discourse analysis was not more resource consuming than for classical qualitative trade-offs assessments, whereas analysis quality increased.

Second, considering in an explicit way ES facets is a relevant step towards a more equal accounting of the social and ecological systems and of their interactions, which in turn is required for adaptive spatial planning (Bennett et al. 2009, Chan et al. 2012, Ban et al. 2013, Karrasch et al. 2014). To date, much more work has been focused on the ecological side than on the social one (Bagstad et al. 2014), and calls have been made to reach better balance between both aspects (Spangenberg et al. 2014).

Third, by explicitly accounting for ES facets in the INF, we considered jointly in the analysis various spatial scales. As an example, agricultural production is supplied at field scale; its demand facet arises from a larger one as products could benefit local people, tourists and more remote populations; and the actual service depends on both the farmer's practices at local scale and on external factors a larger scale (e.g. European and national policies). Thus, considering ES facets is a way to acknowledge that social scales cut across biological boundaries (Hein et al. 2006). Consequently trade-offs and synergies between ES facets also happen at multiple scales and focusing on a single scale would not convey a comprehensive vision of the system. As such, we promote the explicit consideration of the distinct facets of ES and of related scales to support effective management actions (Willemen et al. 2012).

Fourth, by including specifically the actual ES facet, the IFN integrated external variables whose influence could have been overlooked otherwise. This is consistent with Spangenberg et al. (2014) who located pressures (namely “anthropogenic, social and biophysical impacts on biodiversity, ecosystems and their services”) at the interface between biosphere and anthroposphere, which is what is being represented by the actual ES facet. For instance, a positive influence relationship was discussed from actual wood production onto actual leisure hunting thanks to an increased accessibility for hunters by logging roads (Figure 8). This connection between forestry and hunting activities would not have been revealed by a focus on potential supply or demand facets. Moreover, policy was observed to impact only the actual facet in certain cases. For instance, water regulation impacting the hydro-energy service had no influence on potential supply (depending on slope, precipitation and watershed vegetal cover mainly), neither on demand (relying on the social value attributed to renewable local energy). Nevertheless, environmental legislation in the French Alps has reduced actual hydro-energy power supply in order to increase minimum downstream flows.

An interesting follow-up of our analysis would be to mobilise the INF for a more precise analysis on the evolution of emitted/received ratio according to ES categories and facets.

4.4 Governing complex social-ecological systems

While influence relationships between pairs of variable remained simple (Fig. 6 to 9), the leisure hunting example pointed out the rapidly increasing complexity of real systems (Figure 10). Therefore a balance needs to be found to provide graspable although comprehensive information. Many tools can be used to improve knowledge and raise awareness for environmental management and communication. Such tools include participative mental models (Moreno et al. 2014), fuzzy cognitive maps (Kok 2009), bayesian belief networks (Landuyt et al. 2013), social network analysis (Hicks et al. 2013) and, as presented in this article, influence networks.

Finally, in-depth understanding of ES trade-offs and synergies can support the governance analysis of environmental features. This is relevant because trade-offs between ES can be aggravated by conflicting goals of different policy instruments. For instance in Europe, food production supported by the Common Agricultural Policy can conflict with maintenance of water quality pursued by the Water Framework Directive (Hauck et al. 2013). Additionally, the frequent mention of policy as driver of ES interaction in our analyses highlighted the need to relate understanding of ES trade-offs to governance issues, as had been advocated by other authors (Briner et al. 2013). Practical implementation of such governance analysis has been successfully carried out for single ES with participative mental model (Moreno et al. 2014). We anticipate that a main interest of the INF lies in its suitability for, as a next step, mapping policy networks upon ES networks, thus providing innovative and effective understanding of the governance of complex systems.

V. Synthesis

This chapter was dedicated to the qualitative assessment of influence networks around ecological parameters over the French Alps. The new Influence Network Framework (INF) expands on previous methodologies and in particular relates the interests of interaction frameworks and of conceptual developments on ES facets.

Figure 7 below summarizes the framework that guided this analysis as well as the main resulting outputs.

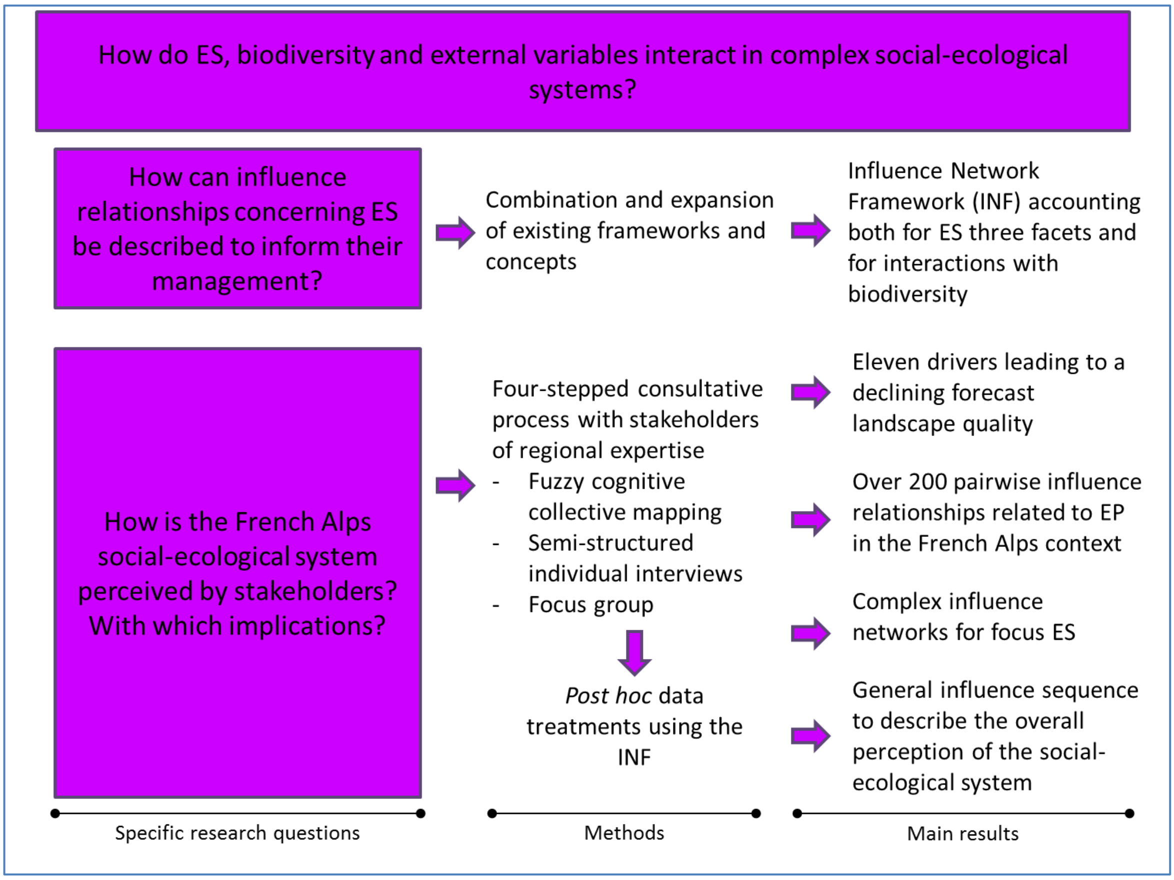

Figure 7: Specific research questions explored in the qualitative assessment of influence networks among ecological parameters (EP) (Chapter II), related methods and main results obtained.

The implementation of the INF for an approach of the French Alps system provided me with the opportunity to encounter various stakeholders. I highly appreciated these meetings, although some challenged me by being rather critical regarding the concepts, methods or objectives we mobilised. Overall, the consultative process presented here has been essential to build my vision of the social-ecological system. It also contributed to the conceptual maturation proposed here as the INF. Finally, at a personal level, I am grateful for these exchanges that widened my understanding of opinions, concerns and perspectives regarding the management of natural resources over the region.

I believe the INF has the potential, as demonstrated here for the French Alps, to foster progress in the understanding and description of complex systems, accounting for varying perceptions of ES relations across spheres (ecological / social), scales (local to global) and opinions (depending on stakeholder groups). I anticipate the interests of this framework as a basis for the choice of relevant management options and governance analysis. Indeed, as further exposed in Chapter III, the INF can describe the influence relationships that need to be managed to sustain the supply of given ES or to maintain environmental quality in general. Then, relevant policy instruments can be additionally presented on the influence networks so as to discuss their interests and limits, individually or in relation with others.

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Chapter III – Policy mix analysis

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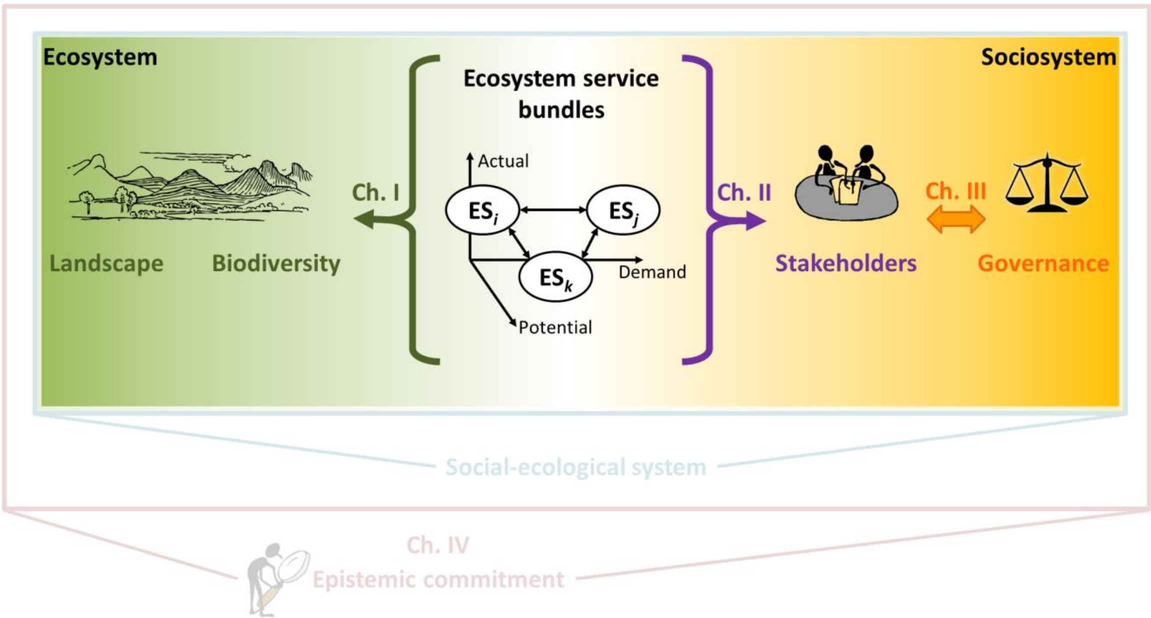


Chapter 3 – Policy mix analysis

Chapter III aims at testing a methodology for assessing the ability of governance to sustain ES supply and to conserve biodiversity. The method has been proposed by our partners from CONNECT project (Universitat Autonoma de Barcelona – Work Package 5) and its implementation in the five case-studies included in CONNECT intends at testing its practical potential. Overall, this method approaches environmental governance through its instruments, and more specifically targets the effectiveness of a policy mix through the information of a large set of criteria. While our assessment of the French Alps system initially focussed on social and ecological features, it appeared interesting to consider additionally the formal set of rules enabling the management of the ES and biodiversity variables we explored. Due to a lack of disciplinary background in governance analysis and also regarding the limited timespan we disposed of, the work I present here is to be taken as a first approach of governance, moreover focused on a restricted aspect of the system (agriculture / tourism / biodiversity) and on a limited number of policy instruments. In other words, the results proposed in this Chapter are not given as a normative judgment on the current alpine governance system. Rather, I propose them as an entry point for discussing i) the interests and challenges of integrating governance analysis in ES assessments in general and ii) some prominent features of the alpine policy mix as we characterised it.

The following sections aim at exploring the policy mix used in the French Alps to manage influence relationships at the interface between agriculture, outdoor tourism and biodiversity. Here, my overarching objective is to increase understanding of influence networks between ecological parameters (i.e. ES and biodiversity) by focusing on the governance instruments currently used to manage them. For this chapter, I worked with a Master student (Elise Trouvé-Buisson – Master 2 Sciences Po Paris) who I co-supervised with Sandra Lavorel during 4 months (September 2014 - January 2015). The results and discussion proposed hereafter come from this fruitful collaboration.

Chapter III is structured in six sections. It does not yet include a paper even though I would like proposing one in the coming months based on the results and discussions presented in this chapter.

- Section I presents the **specific research questions** related to our governance analysis.
- Section II presents the **setting** and justifies our **multi-steps approach**, as we analysed a set of 10 governance instruments relevant for the control of specific influence relationships concerning three domains (agriculture, tourism and biodiversity), chosen among the overall complex policy setting of the French Alps.
- Section III details the **research methodology** we followed and defines the **criteria** we used to analyse the policy mix.
- Section IV rapidly presents our main results regarding **individual governance instruments** and more extensively discusses the **synthetic policy mix analysis**. It includes the **policy brief** we designed to communicate with multiple stakeholders at regional scale.
- Section V discusses the **interests and limits** of our governance analysis and exposes ways of **expanding its scope**.

- Section VI concludes by a **synthesis** of our main insights from this governance analysis.

I. Specific research questions

The overarching objective of this chapter is to test a methodology designed to explore how effective the alpine policy mix is at enhancing biodiversity and ES in the specific context of interactions among agriculture, tourism and biodiversity. I approached this objective through the three following questions:

1) What are the main individual characteristics and rebound effects of 10 policy instruments used to promote or control influence relationships among agriculture, tourism and biodiversity?
2) How are these instruments articulated within the policy network? With which impacts (positive redundancy/negative overlap…)?
3) How can governance analyses inform the management of bundles of ecological parameters (ES and biodiversity)?

To answer these questions, we carried out an extensive review of scientific and expert literature, and further supported it with six interviews with stakeholders of regional expertise. We came out with a set of 10 individual analyses of policy instruments that we further transversally discussed before concluding by producing a policy brief.

Figure 1 specifies the successive steps of this analysis.

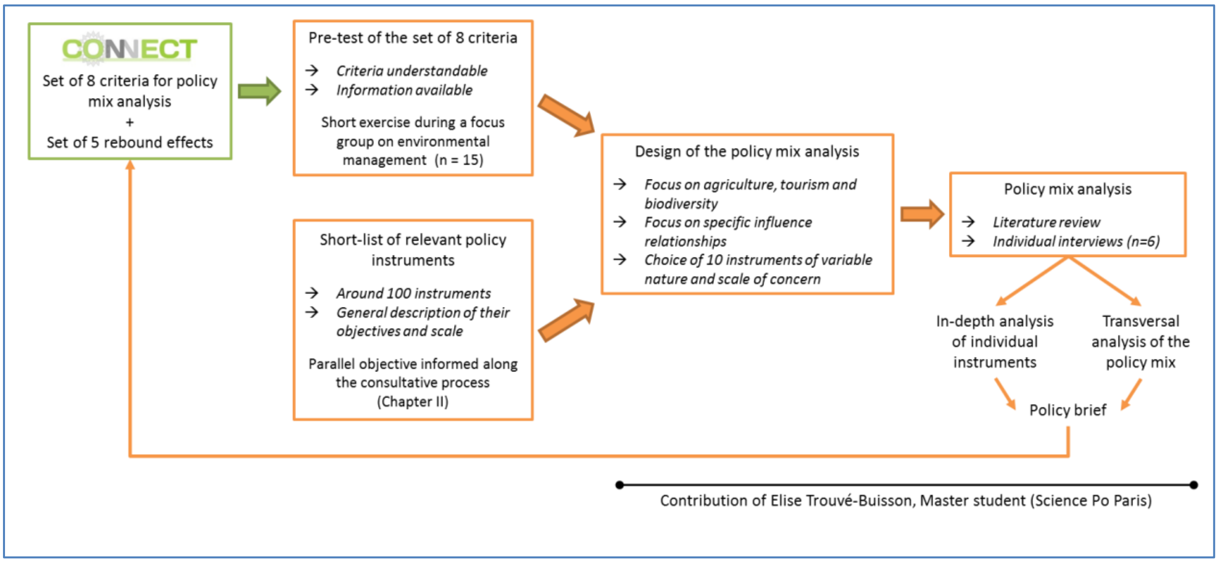


Figure 1: Steps of the policy mix analysis. In green are shown inputs from the CONNECT project, and in orange the work I carried out specifically for the French Alps case-study in the context of my PhD. I acknowledge the rich and fruitful collaboration with Elise Trouvé-Buisson, Master student from Science Po Paris. She was in charge of the policy mix analysis step and performed its synthesis and discussion under my supervision. The outcomes of the analysis presented in this Chapter are thus mostly a collaborative result. They will feedback to CONNECT partners for a synthesis at European scale.

II. Performing a policy mix analysis in a complex setting

A. What is environmental governance – what do we know about it?

"*Environmental governance is varied in form, critical in importance, and near ubiquitous in spread*" (Lemos & Agrawal 2006).

Ecosystems have been used, conserved and restored throughout time based on collective arrangements that enable natural resource management and allocation (Primmer & Furman 2012). The set of collectively acceptable principles that frame these uses is called *governance*. In particular, *environmental governance* refers to "the set of regulatory processes, mechanisms and organizations through which political actors influence environmental actions and outcomes" (Lemos & Agrawal 2006). Governance interests multiple actors, from governmental, inter-governmental, and nongovernmental organisations, from the private sector and from civil society (Greiber & Schiele 2011). Governance induce choosing between multiple options and the "commitment to a particular course of action" (Pielke 2007) is reflected by the formal arrangement laid out by a *policy*.

Two stances traditionally opposed in environmental governance, one seeing in nature (or biodiversity) a source of income and potential uses while the other promoted it as a target for conservation measures (Primmer & Furman 2012). ES have been proposed a relevant concept to go beyond this cognitive dichotomy, in particular by pointing out the importance ecosystem functions and regulating services that were seldom targeted explicitly by governance (Mainka et al. 2005, MEA 2005, de Groot et al. 2010, Harrison et al. 2010). Additionally, authors stressed that objects (i.e. bundles of ES and biodiversity variables) and methods (i.e. multi-dimensional assessments considering ecological, socio-cultural and economic aspects, scenarios, participative approaches) scoped by ES science can be usefully orientated toward the assessment of environmental governance (see for instance Palomo et al. 2011, Lamarque et al. 2014). Addressing environmental issues has been acknowledged a global and critical endeavour that led to a number of political commitments referring to both ES and biodiversity targets (Daily et al. 2008, Carpenter et al. 2009) and the potential of the ES concept for making these commitments more environmentally effective will be tested through time.

While research on environmental governance has a long history, the seminal advances by Elinor Ostrom (1990) on small-scale environmental governance and Oran Young (2002) on international environmental regimes are considered milestones for current works (Epstein 2015). Four themes appear topical in environmental governance research.

1. Influence of scales

Complexity of environmental governance is partly linked to its multiscalar character, because "services generated at a particular ecological level can be provided to stakeholders at a range of institutional scales, and stakeholders at an institutional scale can receive ecosystem services generated at a range of ecological scales" (Hein et al. 2006). Thus, the "decoupling across scales of the causes and consequences of environmental problems introduces major concerns about the unequal distribution of costs and benefits of environmental issues" (Lemos & Agrawal 2006). Assessment frameworks explicitly integrating the scales of ES supply, demand and management have been proposed (e.g. Hein et al. 2006) and empirically tested (e.g. Gomez-Baggethun et al. 2013). Two main insights steam out from these works. First, scales misfits between supply, consumption and control of ES appear to foster environmental conflicts (e.g. Martin-Lopez et al. 2011). Second, multilevel governance, characteristic of what Ostrom called *polycentric and adaptive political systems*, holds great potential to overcome the issues linked to decision-making processes fragmented over sectoral, territorial, social and political divisions (Lemos & Agrawal 2006, Pahl-Wostl 2009).

2. Power relationships

Governance regimes are characterised by the relative influence of various categories of actors, which are usually broadly divided between *state* and *non-state* actors, the latter being further separated between *markets* and *communities* (Lemos & Agrawal 2006, Pahl-Wostl 2009). Influences among stakeholders are conditioned by power relationships, that can be "formal (e.g. property rights, access, or legal permissions), informal (e.g. social leadership, gender inequity), or hidden (e.g. social pressure promoting self-censorship)" (Felipe-Lucia et al. forthcoming). In western democracies, the last decades have been marked i) by a weakened influence of state actors and ii) by the rise of market-based instruments and of participatory approaches in environmental management (Lascoumes & Simard 2011). Thus, a current challenge for governance is on the one hand to understand how the relative influence of various actor categories affects meaningful policy changes and on the other hand to determine the consequences of varying degrees of stakeholder engagement (Ban et al. 2013, Epstein 2015). Methods to identify and characterise stakeholder engagement have been strengthened (e.g. Reed et al. 2009, Pade-Khene et al. 2013) and the ES literature particularly explored the consequences of *power asymmetries* regarding payments for ES (e.g. Kosoy & Corbera 2010, Pirard et al. 2010, Banerjee et al. 2013) and impact on ES flows (e.g. Grard 2010, Felipe-Lucia et al. forthcoming). Two messages arise from these works. First, they highlight the necessity to identify and limit power discrepancy between the stakeholders that manage, use and damage ES in the objective to sustain adaptive capacity in environmental resource management (Pahl-Wostl 2009). Second, *hybrid* mods of governance going beyond the usual categories of actors (including comanagement, private-social partnerships and public-private partnerships) seem to hold higher capability to address current complex environmental problems (Lemos & Agrawal 2006, Lascoumes & Simard 2011).

3. Accounting for social and ecological dynamics

To progress in ecosystem sustainable management, there is a need to deepen the understanding of factors driving the supply and consumption of ES. In particular, authors have called for an increased embedding of social considerations into ecological understanding (Ban et al. 2013). Various frameworks have been proposed to explicit the determinants of actual environmental management. Among these, I propose three examples. First, considerable credit has been given to Elinor Ostrom's "Institutional Analysis and Development framework" (Ostrom 2009) which has been largely used to enhance understanding of the governance processes responsible for uses of and impacts on environmental resources (Ban et al. 2013). Second, another interesting approach of governance is proposed by D. Waltner-Toews under the acronym AMESH, for 'Adaptive Methodology for Ecosystem Sustainability and Health'. This framework describes current ecosystem organization and uses narratives to describe future pathways relevant for managing environmental and health issues. It has proven useful in collaborative approaches carried out mostly in developing countries (Waltner-Toews et al. 2002). Third, mental models have been mapped to elicit the drivers of individual ES, in order to ease their inclusion into management, as exemplified recently from two stakeholder consultations in Andalusia, Spain (Moreno et al. 2014). Overall, all methodologies consider social, ecological and institutional aspects for governance of natural resources. They often include a temporal dimension and integrate feedback loops among variables.

4. Evaluation of success

Assessing whether the governance of natural resources actually provides desirable social and ecological outcomes (OECD 2007) is increasingly attracting the attention of various stakeholders (Epstein 2015). Performance assessments seek to i) design appropriate policy tools, ii) offer guidance among multiple approaches in a given context, iii) rationalise the mechanisms for implementing governance and iv) favour transparency and social learning in a dynamic process (Conley & Moote 2003, Ring & Schröter-Schlaack, 2011, Coreau & Conversy 2014). To evaluate success, many indicators have been developed and studies consider generally "ecological performance (i.e. resource conditions, sustainability), social performance (i.e. livelihoods), and social justice (i.e. participation, equity)" (Pagdee, Kim & Daugherty, 2006, in Epstein 2015). However, defining precisely what a "good" governance is remains complex (Bovaird & Löffler 2003), for at least two reasons. First, there is often a discrepancy between the subjective appraisals of the outcomes by concerned stakeholders on the one hand and on the other hand the 'objective' measures monitored by an outsider (Epstein 2015). Second, generalisation of key features for success is still challenging, as adapting policies to the characteristics of each specific context seems necessary for them to be effective. Indeed, "one-size-fits-all policies are rarely successful" (Basurto & Ostrom, 2009, in Epstein 2015, Young 2011). To date, there remains a need for increased comprehension about "the conditions under which specific policy instruments are likely to prove effective and how to make use of diagnostic procedures to bring this knowledge to bear on specific cases" (Young 2011).

The approach of governance that is proposed in this Chapter relates to the fourth theme exposed above, i.e. the evaluation of governance ability to manage environmental resources.

B. On the complexity of governing environmental issues

Integrating environmental objectives in sectoral policies (e.g. agriculture, transports…) and managing the ES jointly supplied by multifunctional landscapes have been given as key points to progress toward "an 'ecosystem-based approach' to […] sustainable development policy" (EASAC 2009). However, environmental management in general, and biodiversity conservation in particular, remain governance challenges for at least four reasons (Undertal 2010).

First, they require <u>long-term commitments</u> for actions implemented to be effective and to sustainably enhance environmental quality. There is a risk that addressing short term issues prevail in governance, favouring adaptation over mitigation of environmental problems (Lemos & Agrawal 2006).

Second, environmental governance is faced with <u>complex systems</u> relying on nested social-ecological mechanisms of which we have limited understanding (Pielke 2007, Barnaud & Antona 2014 – see Chapter IV). As we have *no analogue state* (i.e. no system of reference) to anticipate the consequences of our decisions (Undertal 2010), environmental governance needs to be flexible, adaptive and innovative.

Third, environmental quality and biodiversity conservation cannot be achieved through any unilateral effort and a <u>collective form of commitment</u> is required. Management of collective goods has been largely discussed and options include, in a debate still alive to date, privatisation, mutual coercion, education or self-organising actions (for two opposed stances see Hardin 1968, Ostrom 2009).

Four, joint maximisation of ES supply and biodiversity conservation cannot be achieved at all scales and over all areas (e.g. Chan et al. 2006, Rodriguez et al. 2006, Crouzat et al. *in review*). As an example, I participated in a comparative assessment of conservation scenarios at EU scale that prioritized either vertebrate diversity conservation or the supply of a set of 10 ES. We assessed the ability of each scenario to additionally protect the other variable (i.e. ES in the biodiversity-orientated scenario and vice versa). Our conclusions were threefold: 1) both scenarios are better than a random pattern of area conservation for the untargeted objective; 2) even within the dedicated scenario, all dimensions are not ideally protected (i.e. biodiversity scenario protects unequally different vertebrate groups / ES scenario protects unequally the different ES); and 3) the biodiversity scenario does a better job overall for sustaining ES than the ES scenario for protecting biodiversity. Overall, this example at European scale confirmed the need to go further than the strict protection of sensitive areas and biodiversity hotspots to sustain environmental quality, in particular by broadening habitat management strategies (see also Anton et al. 2010). I refer interested readers to the dedicated paper (in which I am co-author): Zupan et al. *submitted*. (at the end of the manuscript in the Appendices from Chapter III (Section A)).

Overall, considering these four challenges, there is a need to '*fit'* governance to environmental issues (Undertal 2010). Authors have proposed to favour policy mixes (Ring & Schröter-Schlaack, 2011, Lascoumes & Simard 2011) and hybrid modes of governance (Lemos & Agrawal 2006, Pahl-Wostl 2009, Lascoumes & Simard 2011). This would enable combining elements from i) a traditional model of centralised power offering the means and determination to achieve commitments, with elements of ii) adaptive governance offering more flexibility and enhancing collective social learning (Undertal 2010).

C. Approaching environmental governance through its instruments

While the two first Chapters of this manuscript focused on social and ecological aspects, this third Chapter targets the *institutional arrangements* characteristic of the alpine social-ecological system. In other words, it considers the articulation of "rules governing the behaviour of actors" (Pahl-Wostl 2009) that enables the joint management of multiple ES and biodiversity.

Institutions can be explored to distinguish between formal and informal governance mechanisms. As defined by C. Pahl-Wostl (2009), *formal institutions* are "linked to the official channels of governmental bureaucracies. They are codified in regulatory frameworks or any kind of legally binding documents. Correspondingly they can be enforced by legal procedures". At the opposite, she defines *informal institutions* as "socially shared rules such as social or cultural norms. In most cases they are not codified or written down. They are enforced outside of legally sanctioned channels". Sharing aspects of formal and informal institutions, markets are a governance mode that gained increasing importance in the past decades, echoing the current neoliberal economic paradigm (Lascousmes & Simard 2011). In real systems, environmental governance is exercised through varied institutions that address different dynamics of change (e.g. markets respond more easily to change than formal institutions such as legislation or property rights, the latter being more easily transformed than informal institutions such as traditions, norms and beliefs) (Kingston & Caballero 2008). Recent works show that it is in the diversity of institutions that governance can reach higher adaptive capacity (Pahl-Wostl 2009) and lead to a multifunctional management of ecosystems (Garcia-Llorente et al. 2012).

In this work, I entered governance through the analysis of some of its formal instruments. As thoroughly explained in Lascousmes & Simard (2011), formal instruments are relevant variables to trace changes in the way society addresses natural resource management. They materialise intentions and explicit societal means to deal with these issues, i.e. they represent the '*how*' of environmental management (Simeon 1976). While exploring informal institutions would indeed provide insightful elements (see Section V), formal instruments i) were the target of the methodology we wanted to jointly test across case-studies in the CONNECT project and ii) appeared a simple entry door for governance analysis, supported by official documents and explicit stakeholder arrangements.

Formal policy instruments are usually divided in three categories (table 1).

Table 1: Generic definition and examples for the three natures of policy instruments, as found in litterature. Definitions and examples are quotations from Ring & Schröter-Schlaack, 2011.

	Definition	*Examples*
 **Regulatory instruments**	Directly control or restrict environmentally damaging activities.	*Permits, standard-setting and zoning or planning*
 **Economic instruments**	- Put a price on environmentally damaging behaviour, thus internalising negative externalities. - Reward conservation enhancing behaviour, thereby addressing positive externalities.	-*Environmental taxes, charges and fees* - *Payments for environmental services and ecological fiscal transfers*
 **Voluntary instruments**	Shift individual or community preference functions towards more conservation and inform or educate people about relationships between their activities and the environment.	*Informational and motivational instruments*

Our approach for governance analysis comprised two steps. First, we identified 10 instruments currently proposed to manage bundles of ecological parameters (i.e. ES and biodiversity). Second, we assessed whether the means reached expectations, i.e. whether the environmental objectives were actually achieved or not. Overall, our analysis allowed us progressing in the understanding of how effective alpine governance is for managing a specific bundle of ecological parameters (i.e. agricultural production – nature tourism and biodiversity), from the particular stance of its policy instruments.

III. Research methodology and criteria grids

A. The need to focus on a restricted set of instruments

In the two previous chapters, I exposed the diversity of biophysical conditions and of human uses found in the French Alps. Altogether, they are responsible for a high diversity in biodiversity, ecosystems and ES (Tappeiner et al. 2008, Crouzat et al. in review). Managing any single component of the social-ecological system is demanding, due to the large number of related influencing variables and impacted variables (see Chapter II). The network of policy instruments that was progressively constructed by society to frame the impacts of human activities on ecosystems is therefore highly complex. This network is usually called a "policy mix", defined in this context as "a 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" (Ring & Schröter-Schlaack, 2011). Previous works conducted in my team in LECA highlighted the complex interplay of stakeholders and instruments concerned by ES governance in the specific context of high altitude grasslands (Grard 2010). They concluded that some ES, and in particular regulating ES, were seldom targeted by policy instruments while others ES were well integrated (e.g. provisioning and cultural ES), which was considered as a threat for their joint sustainable supply. They also highlighted a power asymmetry between stakeholders at the expense of the farmers, although these remain the prime users and managers of ES. Finally, they stressed the critical importance of extension organisations as links across governance scales, notably in the context of CAP global reforms and changes.

I first discovered the complexity of the alpine policy mix during the consultative process described in Chapter II. While interviewing stakeholders and conducting our focus groups, I additionally asked about major policy instruments currently used to manage the interactions between human activities and ecosystems. As a result, around 100 'important' instruments were mentioned by stakeholders, mainly regarding nature conservation, urban planning, forestry, agriculture, water management and tourism. Stakeholders described these instruments as highly interrelated and insisted on their nested scales of influence, from European to local. As a pre-treatment for the governance analysis, I rapidly explored the main characteristics of this first short-list of instruments by describing their main objective, domain, scale of application and nature (results not shown).

In the contexts of the CONNECT case-study and of my PhD project, I had neither the capacity nor the objective to carry out the assessment of the whole alpine policy mix. Instead, my objective was to identify and characterize a restricted set of instruments used to manage important relationships from the bundles and influence networks I established previously (Chapters I and II). This restricted set acts as an entry point on the broader policy mix and as a first sample to test the assessment methodology proposed by CONNECT partners. With this approach, I did not aim at concluding on the overall performance of alpine environmental governance but rather at collecting some initial information to decipher the general functioning of the policy mix and the mechanisms of association between instruments. These insights can inform the management of bundles of ecological parameters described previously through biophysical and socio-cultural perspectives. I used three steps of selection to identify the core set of 10 instruments whose analysis was performed jointly with Elise Trouvé-Buisson during her master project.

- The first step of selection concerned the domains on which to focus (i.e. on the sectors of activity / of concern). Due to their economic importance at regional scale and to the magnitude of their impacts on ecosystems (both positive and negative), we decided to concentrate on agriculture and tourism. Biodiversity naturally composed the third pillar of our analysis as progressing in the understanding of interactions between biodiversity and ES was our overarching concern throughout this project (CONNECT objectives). Of course, we could have made other choices and focused alternatively on forestry or water management for instance. However, I believe that this focus is relevant regarding the widespread, diverse and multifunctional landscapes concerned by agriculture and tourism activities in the French Alps (i.e. not restricting us to specific ecosystems such as forests or wetlands).

- Agriculture, tourism and biodiversity share numerous and contrasted influences (Chapters I and II). Our second step of selection dealt with the focus on specific interactions among these three domains. We chose simple and yet important examples among the positive and negative mutual influences they share. Our final selection comprised eight relationships representing important benefits and threats induced by one domain on the other (Figure 2).
- Our third and final step of selection focused on the instruments of our policy mix analysis. Based on the initial short-list I obtained from the consultative process, we identified for each influence one or two instruments currently used to manage it. This selection relied on discussions within the scientific team and depended on the amount of information available to inform the individual analysis (scientific literature and expert reports). Moreover, we paid attention to exemplify multiple scales of influence (European Union – national – regional - local) and natures of instruments (regulatory – economic - voluntary). Our selection is neither exhaustive nor fully representative of the broader policy mix. It comprises usual instruments of widespread use with large impacts on ES and biodiversity (e.g. from the Common Agricultural Policy - CAP) and also small scale pilot instruments of much restricted impact but whose functioning seemed insightful in a broader perspective.

Our final set of 10 instruments will be referred to according the following abbreviations (French name is indicated in italics after the English definition):

- **UTN**: Authorisation for new tourism facilities
  - *Procédure Unité Touristique Nouvelle*
  - Regulatory instrument
  - Derogation procedure from the Mountain Law. The Mountain Law aims at limiting impacts on natural habitats from urbanisation and tourism infrastructures in sensitive mountain areas. The UTN can authorise the development of tourism infrastructures if the magnitude of their impacts remains limited and controlled.
- **SRCE**: Regional scheme for ecological coherence
  - *Schéma Régional de Cohérence Ecologique*
  - Regulatory instrument
  - Land planning document aiming at ensuring ecological connectivity through the maintenance of green and blue corridors at regional scale
- **PTCA**: Tourism protocol of the Alpine Convention, an international treaty whose objective is the sustainable management of the Alps
  - *Protocole Tourisme de la Convention Alpine*
  - Regulatory instrument
  - Legal framework supporting an environmentally-friendly tourism and taking into account the needs of tourists and local populations
- **PNAL**: Wolf national action plan
  - *Plan National d'Action Loup*
  - Economic instrument

    - Collective plan to i) support the adaptation of pastoral management to the presence of wolf, ii) protect and enhance wolf populations, and iii) increase scientific knowledge on wolf species
- **PDR**: Regional plan for rural development
    - *Programme de Développement Rural Régional*
    - Economic instrument
    - Implementation of the Second Pillar of the European CAP and set of measures and premiums chosen by the region
- **PHAE2**: Grass premium from the CAP - second pillar
    - *Prime Herbagère Agro-Environnementale 2*
    - Economic instrument
    - Premium aiming to compensate for the decrease in yields linked to an extensive management of grasslands that is beneficial for the environment and biodiversity
- **IG**: Geographical indications for agricultural products
    - *Indications Géographiques i.e. AOC – AOP – IGP*
    - Voluntary instrument
    - Voluntary identification for an agricultural product as originating from a given region and produced according to certain specifications that ensure its quality. Environmental gain is not the prime objective but is indirectly supported (Lamarque & Lambin 2014).
- **AeA**: Pilot project for tourism diversification in pastoral activities
    - *Alpe en Alpe*
    - Voluntary instrument
    - Experimental support for voluntary diversification of pastoral activities. It is based on the development of tourism offer for discovering mountain grasslands and related farming activities. It targets a public from 'soft' forms of tourism and directly involves the farmers.
- **PAEN**: Protective zoning for natural and agricultural areas
    - *Périmètres de protection et de mise en valeur des espaces agricoles et naturels périurbains*
    - Voluntary instrument
    - Regulatory instrument for the protection and higher consideration of agricultural and natural lands in peri-urban areas, to be used mostly in contexts of strong competition for land
- **ENS**: Protected sensitive natural areas
    - *Espaces Naturels Sensibles*
    - Voluntary instrument
    - Regulatory instrument aiming to protect, manage and open to the public a natural sensitive area.

Figure 2 presents the set of ten policy instruments we chose to analyse, the corresponding eight interactions they contribute to manage, as well as the three domains they address. Relationships among ES and biodiversity are formalised according the Influence Network Framework (Chapter II) pointing out the ES facets concerned by the different influences.

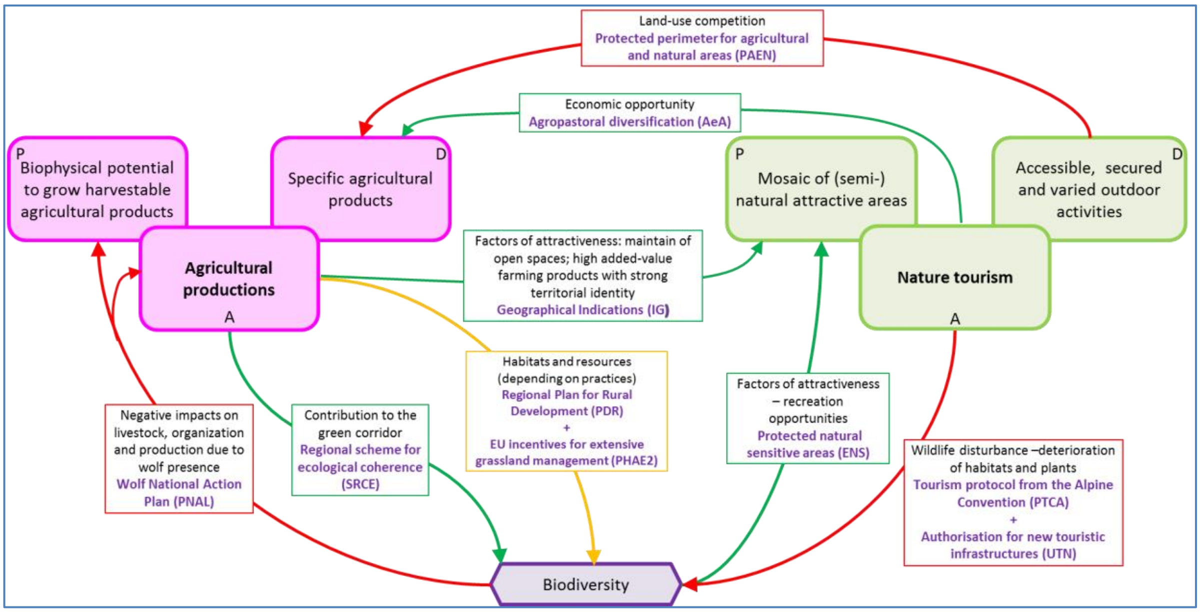


Figure 2: Policy instruments analysed in the French Alps governance analysis (purple text in rectangles). The analysis aimed to address some issues (black text in rectangles) at stake among agriculture, tourism and biodiversity. Interactions are presented as positive (green arrows), depending on practices (yellow arrows) or negative (red arrows). The three facets of agricultural production and nature tourism are symbolised by P for potential supply, D for demand and A for actual supply. For abbreviations of policy instruments, see main text in Section II.

Overall, I designed this study to be relevant for stakeholders of intermediate levels, i.e. at regional and 'départemental' scales mostly. This scope seemed the most adequate regarding the geographical extent of the alpine massif we addressed in our biophysical and socio-cultural analyses (Chapters I and II). Moreover, this scale appears integrative as it articulates broad objectives rising from European and national structures with local needs for practical implementation down to the municipality level. Thus, in short, I believe that addressing intermediate-scale stakeholders is relevant regarding first, the biophysical patterns of congruence between ES and biodiversity we explored, and second, the institutional setting responsible for the French Alps environmental governance.

B. Using the CONNECT grid to assess the performances of individual instruments

Although an objective governance analysis might be unrealistic to achieve, as it usually involves "art as well as science" (Goulder & Parry, 2008, in Ring & Schröter-Schlaack, 2011), numerous criteria have been proposed for the design and evaluation of policy mixes. As detailed in Ring & Schröter-Schlaack (2011), these criteria usually target environmental effectiveness (i.e. the effects of the instrument on environmental quality) and economic efficiency (i.e. the cost/benefit balance linked to the application of the instrument). Further criteria are usually assessed to deal with, among others, fairness, justice, coherence with the legal and institutional systems, or precaution (regarding serious or irreversible consequences that need to be avoided).

For the purpose of our policy mix analysis in the French Alps context, we used a set of 8 criteria proposed by CONNECT partners. This set built on the usual evaluation criteria of effectiveness and efficiency, and required additional information on the instruments' fitting with the broader socio-economic context, on their interactions within the policy mix and on monitoring and control procedures. I am confident that we spanned a wide range of aspects that can affect the final effectiveness of policy instruments, which was our main objective in this assessment. The same set of criteria was used in the policy analysis of other case-studies in CONNECT project, in order to get comparable outcomes that partners from the "Universitat Autònoma de Barcelona", Spain, will synthesize at the end of the project.

To begin with, I conducted a pre-test of the criteria proposed during the focus group described in Chapter II as Step 2. I asked the 15 stakeholders to form groups of 3 or 4 to work collectively on the assessment of one policy instrument of their choice. Our objective was to make sure the list of criteria was understandable and that information on each criterion was available. Outcomes from this experience did not contribute to our final analysis as such but were conceived as a methodological supporting step that we used to compare our theoretical analysis grid with direct expert information. Results were positive and provided interesting information on 4 instruments despite the very short time allocated to this exercise within the focus group program (1/2h). Hence, we kept the initial set of 8 criteria, detailed their definition when stakeholders had asked for more information and further exchanged with CONNECT partners to ensure a common understanding of the assessment grid.

The set of criteria and their final definitions are given in Table 2.

Table 2: Criteria used in the policy mix analysis.

	Criterion	Definition	Question explored
Usual criteria on policy impact	**Effectiveness**	Realization of the environmental aim of the instrument (if the instrument was not designed with a specific environmental aim, we nevertheless evaluated its indirect environmental impact)	- Does the instrument have positive effects on environmental quality? - Is the environmental aim achieved?
	Efficiency	Highest net welfare gain, or lowest net financial cost achieved by the instrument	Is the instrument cost-effective?
	Monitoring and control	Process implemented to ensure that the instrument is applied (obligation of means) or that its objective is achieved (obligation of result)	- Is there a monitoring and control mechanism? - Is it cost-effective?
Fitting with the broader social context	**Equity**	Concept of fair distribution of the outcomes or constraints of the instrument	- Does the instrument guarantee equal treatment for stakeholders? - Who is impacted? Who is excluded?
	Legitimacy	Stakeholder conformity to the process of implementation of the instrument and to its substance-content	Does the instrument appear legitimate to most stakeholders, regarding both its process of implementation and its content?
	Consistency	Good articulation with 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?
	Creation of incentives	Motivation basis on which agents rely to alter their behaviour, e.g. coercion, payment, contract, avoiding a fine/tax...	What drives stakeholders to change the way they act?
Interactions within the policy mix	**Complementarity**	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 others in the policy mix? - Does this combination facilitate the achievement of their objectives?
	Overlap and/or conflicts	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 (e.g. public target, approach) in a policy mix? Is it beneficial or harmful to the overall effects? - Does the instrument conflict with others?

Overlaps are usually defined as negative as they tend to limit flexibility and create unnecessary costs (OECD 2007). However some authors (e.g. Gunningham and Young, 1997 in Ring & Schröter-Schlaack, 2011) consider overlaps to be potentially positive, and point out the interest of redundancies (i.e. positive overlaps) in the particular context of biodiversity policies. Therefore we considered both negative overlaps and positive redundancies in our assessment.

C. Dealing with collateral impacts: the assessment of “rebound effects”

In addition to the ‘classical’ criteria proposed above, we explored the potential ineffectiveness of policy instruments by considering their “unintended, unwanted and avoidable indirect effects”, i.e. the “rebound effects” following the concepts proposed by Maestre et al. (2012). This paper presents a framework for analysing the interdependence between ES, biodiversity and conservation policies. The authors argue that one of the risks faced by environmental governance is to underestimate and thus not anticipate collateral impacts of policies that can undermine their effectiveness and even generate or amplify alternative environmental issues. In Table 3, we propose a short description of the five rebound effects they identified (interested readers are referred to their thorough definition in the original paper).

Although in their initial definition, rebound effects are focused on negative collateral impacts, in our policy mix analysis we considered an extended understanding of this concept. Indeed, we explored also whether the instruments could benefit to untargeted environmental aspects. In the specific context of our policy analysis, we therefore propose both positive and negative rebound effects.

The concept of rebound effect echoes to the awareness that has been rising since the last 30 years in global organisation (e.g. FAO – OECD – UNEP - European Environment Agency) regarding the impacts of public subsidies and tax expenditure on the environment. Several international treaties mention the importance of identifying and controlling the negative collateral effects of instruments For instance, the Strategic Plan for Biodiversity 2011–2020 (Aichi Targets - Convention on Biological Diversity) states that “by 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts” (Strategic Goal A - Target 3). In France, a specific report on public subsidies harmful to biodiversity has been recently delivered to progress on this issue (CAS 2011). However, formal frameworks explicitly accounting for multiple rebound effects are still lacking.

One explicit objective of our methodological testing was to confront the theoretical description of rebound effects from literature analysis with a practical case-study implementation, which has not been done to date. As such, we aimed at identifying the interests and potential limits of this framework regarding both the information available and the insights provided by the assessment of the five rebound effects.

Table 3: Definitions of rebound effects following the framework presented in Maestre et al. (2012). Examples directly come from this paper and therefore concern negative rebound effects only.

Rebound effect	In short	Definition	Example of a negative effect
Biodiversity rebound I	*Spatial spill over* (also called displacement or leakage)	Policy to protect one type of biodiversity in a certain area has an impact on that biodiversity elsewhere, i.e. in another region.	Restricting outdoor recreation in one nature area leads to recreationists moving to other areas so that environmental pressure there increases with potentially negative impacts on biodiversity.
Biodiversity rebound II	*Incongruence or synergy between different types of biodiversity*	Policy to protect one type of biodiversity can affect another type of biodiversity (taxonomic, genetic or functional diversity / rare or common species …).	Providing incentives for habitat protection through creating corridors between protected areas may increase disease risks by promoting contact between wild and domesticated animals.
Ecological rebound	*Impact on ecosystem functioning*	Biodiversity conservation policy might through its effect on particular biodiversity work out negatively or positively on certain ecological relations.	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.
Service rebound	*Trade-off or synergies between biodiversity and ES*	Biodiversity policies can affect positively or negatively the ability of ecosystems to supply services from all categories (provisioning, cultural or regulating).	A trade-off appears between conserving certain species that need dense, old-growth or primary forests, such as the boreal owl, and provisioning ecosystem services, like grazing and timber production.
Environmental rebound	*Shift from one to another environmental problem or solving another environmental problem*	Biodiversity policy can generate a negative impact on certain environmental indicators. Conversely, addressing one environmental problem can contribute to solving another one.	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.

D. Material

We informed the two sets of criteria presented above (CONNECT criteria and rebound effects) firstly through an extensive literature review. Without claiming exhaustiveness, we did our best to consider diverse sources of information (i.e. both academic and expert literature) and paid attention to include the diversity of opinions and judgments expressed by various stakeholders regarding each instrument. As a second step, Elise Trouvé-Buisson, the Master student who assisted me in this analysis, carried out six individual semi-structured interviews. We designed the interviews to validate our literature analysis and eventually to refine it by adding information from important reports we would have missed or from alternative points of view that would not have been expressed in the documents we consulted.

We conceived these interviews as an opportunity to assess gaps between theory found in literature and opinions based on on-the-ground experiences. Due to tightly constrained time availability, our interview sample remains very limited and therefore potentially biased by normative information representing personal opinions from the stakeholders we consulted. We tried to overcome this problem by consolidating through additional literature exploration the new inputs interviewees provided. Each interviewee was asked questions specifically on one or two instruments, as shown in Table 4. Only the PDR was not explicitly the focus of one interview but many of its measures were discussed together with the PHAE2.

We gathered a huge amount of information thanks to the literature review and the interviews. We progressively synthetized it until a final broad assessment on each criteria was obtained. When answers to one criterion included contrasted opinions, we kept this information by a negative assessment (i.e. if some stakeholders judged the equity criterion negatively and others positively, our assessment was negative and highlighted diverging opinions). Even if we tried to keep as much precision in our analyses as possible, we warn against a too strict understanding of the final synthetic judgment provided in section IV and encourage interested readers to consult the more detailed analyses proposed in the final report of Elise Trouvé-Buisson (Trouvé-Buisson 2015). Additionally, I repeat that our objective was not an exhaustive assessment of the alpine policy mix but rather a first approach of important characteristics of some of its instruments so as to test an assessment methodology. Thus, although our assessment are provided as strong statements (i.e. either a positive or a negative assessment of each criterion), I do not pretend having integrated all the complexity of the stakeholder interplay and of articulations with other instruments and institutions that alone would enable proposing a more objective and robust assessment.

Table 4: Number of supporting references (reports, papers, opinion papers…) consulted from expert and academic literature to assess each instrument (detail available at the end of the manuscript in the Appendices from Chapter III (section C)) - Structure and position of the interviewees consulted for validating and completing their individual analyses.

Instrument	Supporting references	Organisation	Position of the interviewee
UTN	6	CIPRA (NGO for the Alps protection and sustainable development)	President
PTCA	10		
SRCE	7	Rhône-Alpes regional environment and agriculture directorate (DREAL)	'Sustainable development and biodiversity' team leader
PNAL	12	Ecrin National Park (PNE)	'Agriculture' park officer
PHAE2	14		
AeA	4	Extension organisation for a sustainable alpine agriculture (SUACI)	'Tourism & Agriculture' project officer
IG	18		'Territorial dynamics' team leader
ENS	14	General Council Isère (CG38 – local government at département level)	Team 'Environment' (*2)
PAEN	11		
PDR	15	-	-

IV. Main individual results and transversal analyses

A. Individual analysis following CONNECT criteria

In Supporting Information (at the end of this manuscript, in the Appendices from Chapter III (section B)), three tables propose our synthetic assessment on each CONNECT criterion for the 10 policy instruments we thoroughly assessed (Table S1: regulatory instruments; Table S2: economic instruments; Table S3: voluntary instruments). Detailed tables with supporting references are available in the final report of Elise Trouvé-Buisson (2015).

Below, I propose three schematic visions to synthetize our results from the individual analysis of policy instruments following CONNECT criteria (Figure 3).

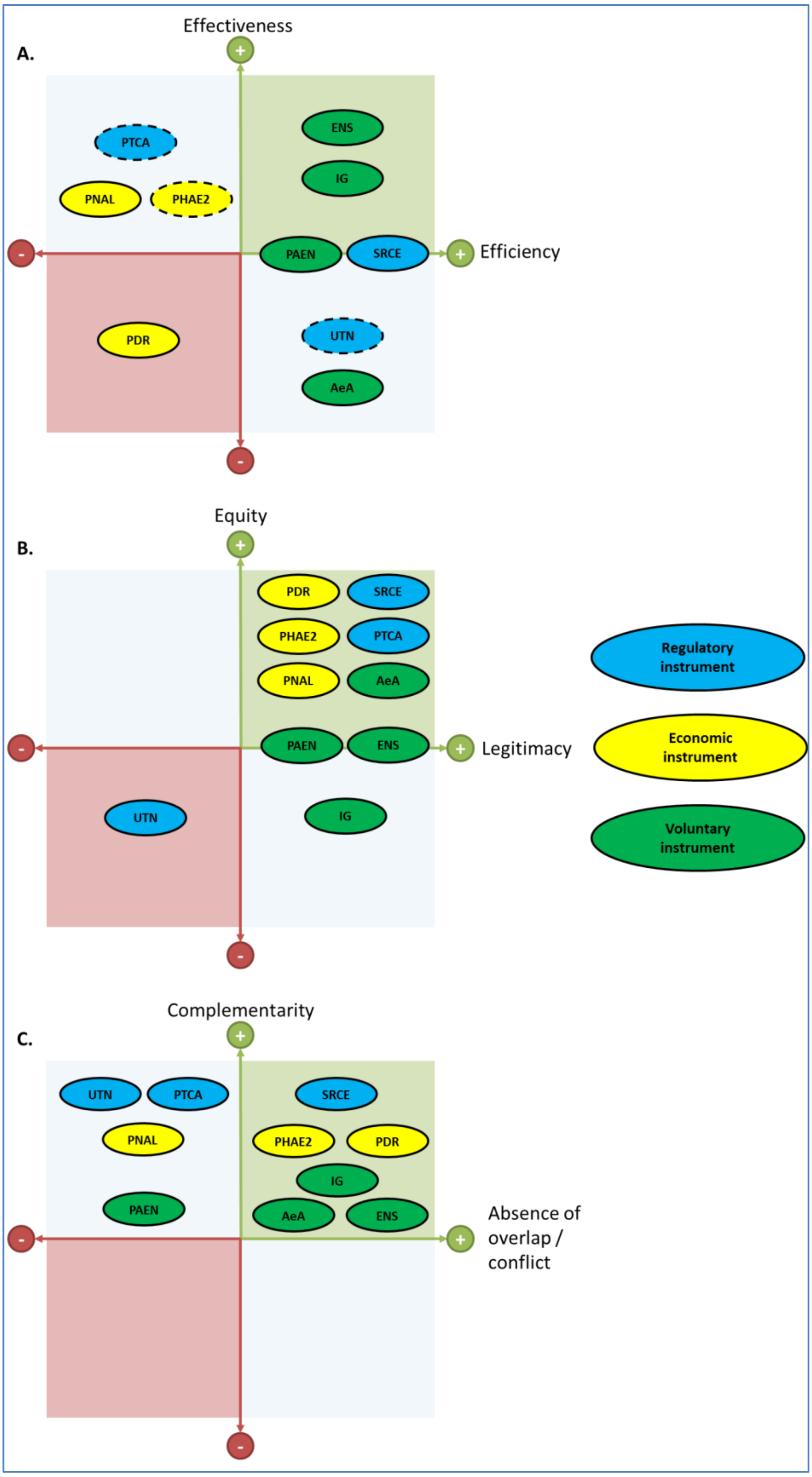


Figure 3: Individual characteristics of the ten policy instruments analysed following CONNECT criteria. Nature of each instrument is shown in blue background for regulatiry instruments, in yellow for economic insruments and in green for voluntary instruments. A. Assessments on Effectiveness and Efficiency, dotted outlines indicate an additional negative judgment on the Monitoring & Control criterion. B. Assessments on Equity and Legitimacy. C. Assessments on Complementarity and Absence of overlap / conflict.

I highlight below some important features summarising the characteristics of the policy mix following CONNECT criteria.

Three instruments performed negatively regarding the environmental effectiveness criterion (Fig. 3A). Reasons invocated to justify this judgment differed: a restricted scale and no direct environmental objective for AeA, little actual environmental gains and stakeholder collaboration for PDR, and an instrument being by essence a derogation procedure from a more conservative strategy (Mountain Law - '*loi Montagne*') for UTN. At the opposite, we point out the effectiveness of three instruments that demonstrated an actual and widespread support for mountain farming that positively impacts environmental quality (PNAL – PHAE2 – IG).

In Figure 3A, the economic efficiency of two planning instruments for nature protection was assessed as high (ENS - PAEN) as they offer a perennial environmental protection on areas undergoing human pressure. Their cost-benefit balance was thus positive at mid- to long-term. AeA and IG presented a good efficiency as their budget is very limited. This contrasted with four instruments which we assessed as not cost efficient (PDR, PNAL, PHAE2, PTCA). Indeed, they rely on substantial budget (e.g. 10 millions €/year for PNAL at national scale for protection measures and compensations for impacts of a single species – 79.2 millions € for the 2007-2013 PHAE2 program in the region Rhône-Alpes). Additionally, PHAE2 mostly supports already existing practises thereby not creating additional environmental gain, i.e. presenting a "lack of additionality" (Santos et al. 2014). Overall, we warn against a too strict understanding of our efficiency analysis, which negatively weights high net budgets dedicated to single instruments. To go a step further and to be able to explicitly assess efficiency, the policy analyses would need to focus instead on marginal costs and benefits (OECD 2007). This means that the actual cost of an instrument should be compared to the environmental gains or losses it directly induces. Due to lack of adequate data, we were not able to use these marginal criteria in our analysis, whose results remain therefore restricted. A solution proposed to assess marginal costs and benefits of instruments would be to introduce scenarios. By making the policy mix vary, they would assess the marginal effects of the introduction (or suppression) of individual instruments on environmental variables.

We draw attention to the perceived under-optimal monitoring and control procedures for three instruments (UTN – PTCA – PHAE2) (Fig. 3A). In particular, the UTN procedures of control exist but some stakeholders fear that they are sometimes by-passed. Thus, they criticise the instrument for a lack of transparency of its environmental assessments. Monitoring and control procedures have been proven essential to ensure legal compliance, to facilitate adaptive management of individual projects and to provide evidences on the effectiveness and costs of particular measures to all stakeholders concerned, including scientists and decision-makers (Ring & Schröter-Schlaack, 2011).

Equity (Fig. 3B) was positively assessed for all instruments in our policy mix, in particular for those supporting alpine agriculture (PDR – PHAE2 – PNAL – AeA) as they compensate for the additional constraints farmers face in mountain areas. The only two exceptions concerned the UTN procedure whose high costs are restrictive for small municipalities, and the IG which openly promotes differentiation of agricultural products and therefore does not treat equally all farmers. PAEN and ENS were not sanctioned by the equity criterion as on the one hand they restrict some land uses (e.g. urbanisation) and thereby exclude some stakeholders (i.e. deny their private interests, especially regarding urban and infrastructure development), but on the other hand they tend to reinforce global equity by keeping these areas publically accessible and in good environmental condition.

Legitimacy (Fig. 3B) was also positive for all instruments, and in particular for those tightly linked to a participative process (e.g. SRCE). The only exception concerned the UTN, whose impartiality was questioned by some stakeholders in relation to a perceived lack of information on the actual elements of justification for positive or negative derogatory decisions. Overall, legitimacy reflects subjective and personal perceptions, which made its assessment challenging. Therefore, we warn against a too strict understanding of our synthetic assessment. We feel that the positive views which were expressed may have been driven by the widespread discourse on the necessity of nature conservation. Current debates mostly focus on implementation (e.g. specific location, management practises) or on budget allocation.

All instruments were assessed as consistent with the alpine institutional and socio-cultural setting (not shown in Figure 3). As for legitimacy, we warn against a generalisation of this assessment because cultural consistency remains rather subjective. We mostly analysed instruments recently introduced (e.g. the tourism protocol of the Alpine Convention was adopted in 2006 in France) or adapted (e.g. that last modification of the UTN procedure dates back to 2006). These are therefore likely to be well designed regarding the broader policy mix with which they are articulated.

Consistent with our initial approach of the policy mix (consultative process – Chapter II and results from Grard 2010), we found that all instruments presented many complementarities with instruments of diverse natures and related to various scales (Fig. 3C). We thus confirmed that complexity and interconnection characterise our policy mix. I suggest that this situation is not mere chance and make the hypothesis that it stems from the objectives of multifunctional ecosystem management that are common in the French Alps, as reflected in mosaic landscape patterns and specific strategies for agriculture and forestry (Tappeiner et al. 2008, Crouzat et al. in review).

In contrast to this overall positive assessment of complementarities, we found a variety of patterns regarding overlaps and conflicts. Instruments supporting alpine agriculture presented little overlaps and conflicts with other instruments, as maintaining agriculture in these disadvantaged areas can be considered a common endeavour widely addressed (e.g. by the Interregional Convention for the Alpine Massif (CIMA), by the 'Agriculture' protocol of the Alpine Convention…). This positive situation appears strengthened by a careful design in measures and premiums related to the CAP's second pillar (PDR – PHAE2) (EC 2013). SRCE was characterised by positive redundancies with zoning for protection at lower scale. ENS also presented positive redundancies with other small-scale protected areas, including with PAEN. However, PAEN appeared to negatively overlap with specific protective status for agricultural areas undergoing artificialisation pressures (ZAP). This was found confusing by some stakeholders and limits the use of the PAEN instrument to date. The UTN procedure was also assessed negatively assessed as it overlaps with many other instruments. We hypothesise this is linked to the complexity of the alpine policy mix (and further of the French one) regarding urban planning procedures, which is usually described as an 'administrative layer cake' to symbolise the complex interplay between overlapping competent authorities from diverse scales and the related supporting policy instruments they use (Blaise et al. 2003). The negative assessment of UTN is also linked to its very nature of derogation procedure, because by definition it enables artificialisation of sensitive areas which opposes with the conservation objectives supported by other instruments (although the UTN proposes a 'controlled' artificialisation). In addition, PTCA conflicted with other instruments as its broad objectives of sustainable and environmental-friendly tourism conflicted with local preferences for ski resort development. For instance, the procedure for increasing the artificial snow capacity of a resort is only submitted to declaration procedures (and not to authorisation procedures), while the form of tourism it supports is contradictory with PTCA recommendations. Finally, our negative assessment for PNAL is mostly based on conflicts with other instruments. For instance, in areas where predation on herds is high, shooting wolves can be authorised despite the strictly protected species status established by the Berne Convention (1979). Moreover, as highlighted by multiple stakeholders during our assessment of influence relationships (Chapter II), one protection measure supported by the PNAL is the presence of specific protection dogs (‘*patous*’) whose encounters generate conflicts with tourists and hikers. This situation decreases the attractiveness for tourism and recreation in some pastoral areas, which directly conflicts with the objectives of tourism-related instruments such as AeA. Overall, our negative judgment for PNAL is characteristic of the critical tensions linked to the ‘wolf debate’. Indeed, the objectives of i) supporting its recolonisation in the Alps and ii) supporting extensive pastoralism and related environmental benefits are presented as hardly compatible, leading to a fundamental discrepancy between instruments focused on a single one of these objectives. The ambition of the PNAL is to reconcile both objectives and I suggest that it is therefore a very precious, although challenging, instrument of the policy mix.

B. Individual analysis following a rebound effect analysis

In addition to classical criteria for policy mix analysis, we informed collateral effects of our policy mix following the rebound effect framework (Maestre et al. 2012). Analysing these rebound effects is indeed an originality from the CONNECT project regarding the criteria usually proposed to assess policy mixes. As a consequence, our case-study permits testing this novel framework. An outcome of these tests may be that proper applications are challenging and require dedicated in depth analyses.

In Supporting Information (at the end of this manuscript, in the Appendices from Chapter III (section B)), three tables propose our synthetic assessment on each rebound effect for the 10 policy instruments we thoroughly assessed (Table S4: regulatory instruments; Table S5: economic instruments; Table S6: voluntary instruments). Table 5 summarises these negative and positive untargeted environmental consequences.

We often did not find explicit information on rebound effects and our assessment mostly relies on i) hints in environmental assessments sometimes mentioning collateral effects or complementary interests of the instruments, and ii) discussions with the experts consulted whose knowledge of actual consequences of their implementation was highly informative. Overall, we propose a first description of potential rebound effects that should not be understood too strictly as a lack of scientific data prevented us from conducting a truly evidence-based assessment. However, I believe our result is reliable enough to warn against important negative side-effects and to indicate potential synergies between objectives, as developed below.

Table 5: Synthesis of potential rebound effects for individual policy instruments. Red backgrounds highlight negative effects, green backgrounds indicate potential synergies between the objective of the instrument and the component specified by the rebound effect, and orange backgrounds represent variable rebounds whose influence depends on implementation modalities or which are uncertain to date.

	Biodiversity Rebound I	Biodiversity Rebound II	Ecological Rebound	Service Rebound	Environmental Rebound
UTN					
SRCE					
PTCA					
PNAL					
PDR					
PHAE2					
IG					
AeA					
PAEN					
ENS					

The instruments we assessed generate numerous spatial spill overs (Biodiversity rebound I), linked to two main reasons. First, 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 (IG - PDR – PHAE2). Second, the protection of specific areas and their withdrawal from land planning opportunities could increase land pressure on remaining areas that are also potentially of interest for biodiversity (SRCE – PAEN - ENS). However, the relatively small scale of restrictive perimeters (PAEN and ENS) moderates this judgment, as well as the fact that the SRCE heavily relies on already planned protection perimeters (e.g. Natura 2000 and nature reserves). In addition, the stakeholders we consulted mentioned that the UTN could sometimes negatively impacts ecosystems remotely. In particular, new facilities for artificial snow withdraw water volumes and alter the annual water cycle. This disequilibrium can affect downstream biodiversity, so the UTN procedure needs to be carefully designed to account for spatial dependencies. Finally, we highlight one positive spatial rebound (PAEN). One side-effect of protecting agricultural areas in valleys undergoing high urbanisation pressure is to support alpine farms in general and in particular their activities in high altitude pastures. Indeed, available agricultural space in the valleys determines herd size, while these same herds are responsible for maintaining open landscapes at high altitude during summer. Thus benefits for biodiversity at higher altitude are conditioned by the conservation of agricultural land in the valleys. One alpine farmer with whom I discussed this issue during a meeting estimated that withdrawing one hectare of agricultural lands at low altitude was responsible for the abandonment and progressive closure of three hectares in altitude in the medium term. Even if no data is available to confirm this ratio, the trend seems interesting enough to be highlighted.

Rebound effects concerning other facets of biodiversity (biodiversity rebound II) were not straightforward to assess, as we found that all alpine policies focus on specific species or facets of biodiversity. Some instruments focus on iconic species for prioritizing areas to protect (PAEN – ENS – PTCA) or naturally benefit more to species whose habitat is promoted by the instruments (i.e. open agricultural habitats for PDR and PHAE2 in areas that could naturally be covered by forests). Similarly, the concentration of herds in secured areas proposed as one protective measure of the PNAL increases trampling and overgrazing, which is known to alter biodiversity and favour more generalist species (Tasser & Tappeiner 2002). In addition, the SRCE could favour species with strong dispersal abilities that would benefit from the green and blue corridors to colonise new ecosystems. To date, this remains a hypothesis mentioned by the stakeholders we consulted and a current debate in the scientific literature (see Anderson & Jenkins 2006, and Hilty et al. 2006 in Worboys et al.2010). We highlighted three positive rebound effects: both PDR and PHAE2 support functional diversity e.g. by increasing floral resources in extensive grasslands favourable to pollinators, and IG explicitly supports livestock phylogenetic diversity by promoting local and traditional breeds and varieties.

We additionally explored whether the actual implementation of instruments could impact ecological functions (ecological rebound). We found simple examples of such situation, as for instance one direct effect of the UTN procedure is to artificialize ecosystems, which is negative for ecological functioning in general. Moreover, we found evidences in the literature that a lack of coherence in the supply chain for products with IG could negatively affect ecosystems (Lamarque & Lambin 2015). Our assessment was more moderate for other instruments. In particular, we remained uncertain regarding the impacts of an increased connectivity favoured by SRCE on ecosystem functions as a result of colonisation by for instance invasive species (e.g. on the balance between species in vegetal communities). Moreover, depending on management, public over-use in ENS would be negative for ecological functioning (e.g. by over trampling), but attention to this threat seems high as both objectives are explicitly targeted by the instrument. We stressed three positive rebound effects: two are linked to a support of natural ecological dynamics in agro-ecosystems (PDR - PHAE2), and one concerns the increased abundance and role of wolves in trophic networks (PNAL), known in the literature as a 'trophic cascade effect' positive for natural regulation of species abundances (Ripple & Beschta 2012).

Although Maestre et al. (2012) warned against a "fundamental incongruence" between ES and biodiversity, our analysis highlighted only two clear trade-offs with ES. The first one concerned instruments supporting extensive agricultural practises with the inclusion of environmental constraints in management, which usually decrease provisioning services (PNAL – PDR – PHAE2). 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. Therefore maintaining agriculture is a trade-off for these services (e.g. carbon sequestration, maintain of water quality…). The second trade-off was negative for cultural services as the consequences of wolf return and adapted agricultural practises (PNAL) conflict with leisure hunting and recreation activities in higher altitude areas and also tend to impact landscape aesthetic quality. In three cases, we were not able to determine the dominant trade-off or synergies among ES categories because they depend on local management modalities (UTN – ENS – SRCE). All other rebound effects we found regarding ES were positive and stress numerous potential synergies both between ES categories and between ES and biodiversity. In particular, cultural services were supported by all instruments, highlighting the potential for policy instruments to promote this side-effect. Regulating services were also frequently favoured as indirect effects of better environmental quality (ENS - PAEN) and extensive agricultural practises (PDR – PHAE2 – IG). Finally, three instruments explicitly supported provisioning services, although over restricted spatial extents (IG – AeA - PAEN).

Overall, as mentioned in Maestre et al. (2012), environmental rebounds "involve 'invisible' behavioural and economic mechanisms" that are most challenging to detect. Therefore, we insist on the low reliability of our assessment of this criterion, which I nevertheless presented so as to indicate the need for additional data and methodological insights for progressing in its assessment. We did not identify any positive rebound effects, and finally proposed two negative (although uncertain) rebound effects. First, as the UTN is a derogation from the more conservative Mountain Law authorising tourism infrastructure development, we hypothesise that this UTN procedure could increase the number of visitors in the Alps, thereby inducing an increase of greenhouse gas emissions, CO2-intensive energy consumption and water pollution etc. However, we doubt that only the UTN procedure could be mentioned as responsible of these consequences. Second, the decrease in food yields induced by the PHAE2 could be compensated by imports of forage that would induce spatial environmental rebounds and greenhouse gas emissions.

C. Transversal analysis of relationships within the policy mix

Figure 4 proposes a schematic vision of interactions between instruments as explored during this policy mix analysis. It includes the ten instruments we focused on and additionally represents their most important links with other instruments regarding their environmental impacts.

The three clusters we highlight are to be understood as perspectives on the policy mix that should help addressing its interests and limits. In other words, our description of the articulation of policy instruments does not rely on an independent hierarchical classification but rather exemplifies synergies and conflicts or overlaps that are illustrative of alpine environmental governance.

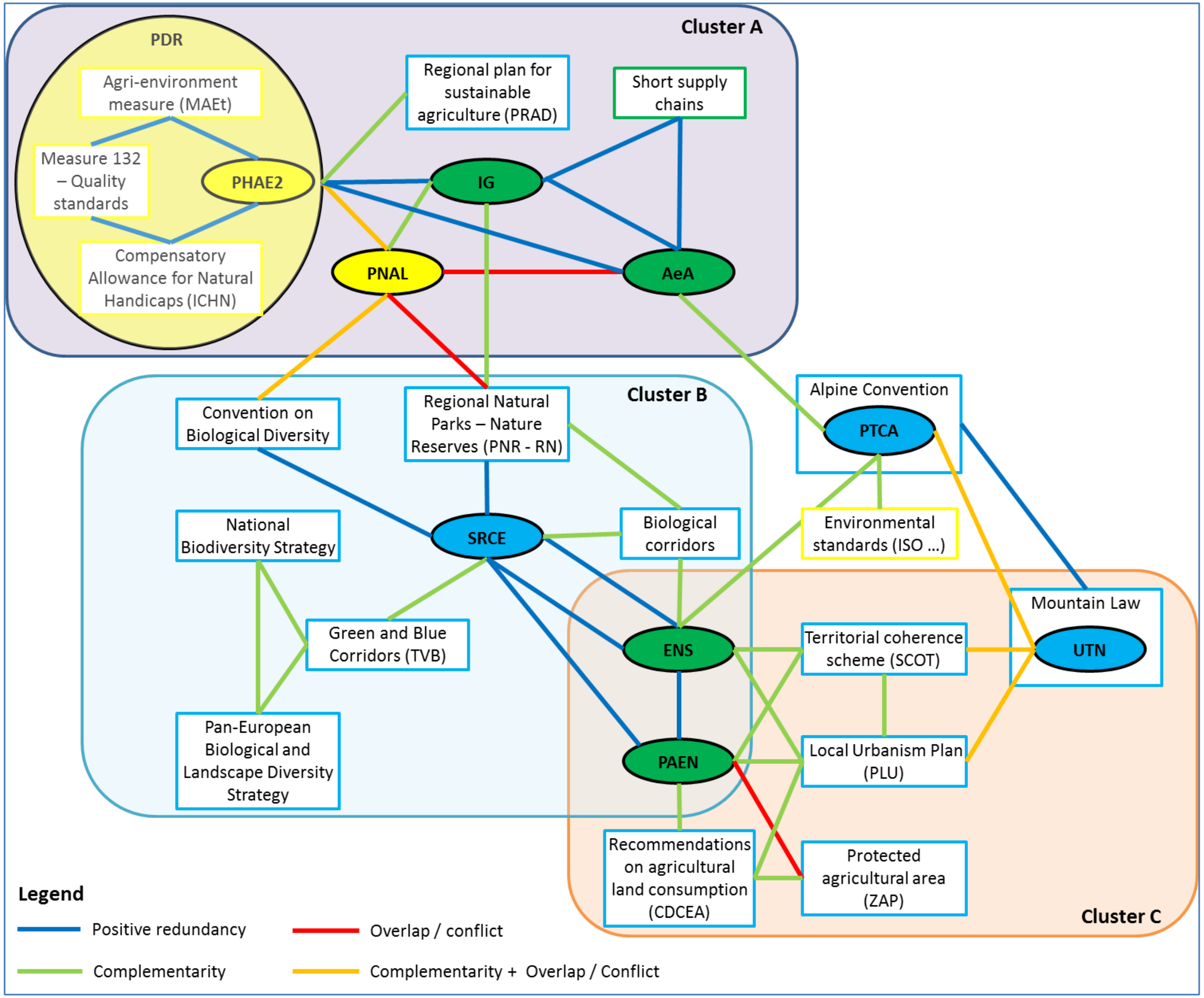


Figure 4: Main interactions between instruments assessed in the policy mix analysis (ovals) and related additional others (rectangles – French acronym in brackets). Three clusters of instruments are distinguished. Nature of instruments is represented in blue for regulatory instruments, yellow for economic instruments and green for voluntary instruments. Four relation types were identified: complementarity (light green), positive redundancy (dark blue), overlap or conflict (red) and both complementary and overlap/conflict (orange).

We identified a first group of instruments targeting maintain of agriculture in disadvantaged areas (Cluster A). This cluster is characterised by numerous positive redundancies as all measures proposed target the same public, i.e. alpine farmers with extensive practises. Their overarching objective is to help maintaining the environmental externalities produced by alpine agriculture and by extensively managed grasslands and pastures in particular. Indeed, the agro-ecological approaches that can be broadly related to alpine agriculture have been proven to favour "'planned' and 'associated' biodiversity in farming systems and agricultural landscapes" (Tscharntke et al. 2012). In addition, grasslands and high pastures of the Alps "play a key role in the agricultural economics whilst being areas of major ecological value" (Noury & Poncet, 2013). However, alpine agriculture faces many biophysical constraints inducing higher costs of production and constraints for mechanisation and the organisation of the work, all of which can lead to agricultural abandonment (Agreste 2013). Therefore, if society choses to benefit from the products and from the environmental externalities of mountain farming, economic, infrastructure, technical and social support are needed. Different instruments can serve this purpose. Among these, we highlight i) instruments to diversify income sources through agro-tourism projects (e.g. AeA), ii) economic incentives conditional on environmental-friendly practises or compensating for external constraints (e.g. PHAE2, PRAD or PNAL), iii) instruments to add value on territorial productions (e.g. IG) and iv) instruments reconnecting producers to consumers locally (e.g. short supply chains). Overall, the instruments in cluster A fight rural decline and increase the long term resilience of agrosystems (Noury & Poncet 2013), which directly benefits alpine biodiversity associated with permanent grasslands and supports the supply of multiple ES from all categories (as found in Chapter I: crop and fodder production – recreation – maintain of water quality and biological control).

Cluster B comprises of instruments fostering ecological connectivity. We believe it represents an archetypal example of a sound articulation of instruments among scales, a characteristic required to achieve environmental gains (Undertal 2010). Indeed, the international agreement on the need to protect biodiversity (Convention on Biological Diversity CBD - 1992) was explicitly inscribed in European policy objectives in the Pan-European Biological and Landscape Diversity Strategy (1995). At national scale, Grenelle's laws I and II (2009/2010) translated these objectives into a framework for the consideration of Green and Blue Corridors ('*Trame Verte et Bleue - TVB*'). Following a decentralised process, regions are progressing in the adoption of their regional scheme of ecological coherence (SRCE) which is the instrument designed to achieve convergence and coherence between upper-scale policy objectives and local-scale governance instruments. As such, we pointed out multiple positive redundancies connecting the regional instrument (SRCE) to both upper- and lower-scale instruments (e.g. CBD at upper scale and ENS at lower scale). The final level concerned local instruments responsible for the actual operationalization of the ecological network that include both hotspots of biodiversity (e.g. ENS, Natura2000) and areas necessary for wildlife mobility (e.g. the 'Biological corridors' program in the département Isère).

The third cluster we highlight is composed by instruments that regulate land planning (Cluster C). It includes numerous local-scale instruments as competence for land allocation is held by municipalities and community of municipalities in France (e.g. PLU – SCOT). Both positive and negative interactions among these instruments were found, reflecting the social synergies and conflicts among sectors of activities and groups of actors. Three factors challenge the effectiveness of instruments to compromise between various land uses. First, land allocation instruments can deal with conflicting objectives. For instance, they need to combine broad objectives of upper-scale instruments as the Mountain Law ('*Loi Montagne*') with the issues decision-makers face at local scale and during their mandate (i.e. in the short term). Hence, new equipment for artificial snow can be locally supported for economic reasons and be designed so as to pass through the UTN procedure, and yet conflict with broad objectives of the PTCA which supports a less intensive form of outdoor recreation. Second, the complex *opposability requirements* (i.e. the authority sequence among instruments that create a hierarchy between their objectives) that link instruments at nested scales challenge their effectiveness. Regarding (again) tourism infrastructures, compatibility is required between local planning documents (PLU – SCOT), between these and the frame objectives defined in the Mountain Law (UTN), and additionally there is a need to account for documents of higher level (PTCA). While we acknowledge the necessity of precise opposability requirements to ensure policy coherence and to enhance enforceability of broad objectives, we also stress the complexity of dealing with such an administrative layer cake. Third, similar objectives targeted by instruments addressing the same scale can lead to confusing overlaps that lessen policy legibility and impacts. For instance, we highlighted an overlap between two protective perimeters at local scale for agricultural areas undergoing artificialisation pressures (PAEN - ZAP) which currently results in an under-optimal mobilisation of PAEN, despite the perennial protection it confers to the environmental externalities of agrosystems.

D. A policy brief to communicate on environmental governance

To conclude our governance analysis, we delivered in a policy brief our recommendations for an increased environmental consideration in the management of agricultural, tourism and natural areas (Figure 5 – in French). It was designed for stakeholders at intermediate level (i.e. regional and départemental scales) as i) they correspond to the level of our biophysical and socio-cultural assessments (Chapters I and II) and ii) these are levels that appeared important to consider from the analysis of our governance results.

We propose this brief as an integrative analysis relying on policy relevant ecological and socio-cultural knowledge. Producing this brief was part of the expected deliverables in the context of our participation to the CONNECT project. More specifically it was intended to draw policy recommendations from the assessment of the policy mix through the combined set of CONNECT criteria and rebound effects. As our results may not be robust and precise enough to actually deliver realistic and practical policy recommendations, this policy brief rather highlights some key messages that remain general but stress some important trends that were figured out throughout our analysis. The messages we deliver reflect the results of our literature analysis and stakeholder interviews. From all information gathered, we selected key points that either appeared repeatedly or conveyed insightful information from both the opinions of the stakeholders consulted and our experience.

The brief highlights one important challenge faced by each of the three domains we explored (biodiversity – tourism – agriculture) to sustain ES supply and biodiversity conservation and offers suggestions for increasing the environmental effectiveness of their governance. Examples of policy instruments relevant to address the issue are proposed. Additionally, academic research is explored as a fourth domain to propose suggestions for possible improvements in governance studies.

Stakeholders concerned by <u>biodiversity conservation</u> are faced with one important challenge which is to optimize the articulation of policy instruments among political scales. This is indeed required to foster coherence and cost-efficiency of policies. We emphasise the importance of the regional level in this endeavour as a necessary conveyer of information both down to the municipality level and up to the UE level (see for instance the role of SRCE in articulating European and national objectives toward their operationalization through local land allocation instruments). In addition, our analysis conveyed that dialog among stakeholders and across institutional scales would enable higher environmental consideration if it was more transparent, permanent and independent from political mandates. This would enhance both enforceability and social acceptability of environmental governance.

Concerning tourism sector, one important challenge to undertake is to (re)frame alpine tourism projects so as to better align with objectives of sustainable development. This is currently required to limit the environmental impacts of tourism infrastructures and activities. Support should be maintained and strengthened toward forms of "soft tourism" that are carefully adapted to the alpine sensitive setting, including collective transportation and agrotourism projects. Additionally, renovation should be favoured upon new facilities when possible. Based on our literature search and interview outcomes, we recommend a higher transparency of decision processes for authorisation procedures of tourism infrastructures (such as UTN) and encourage an increased mobilisation of framing instruments such as the Alpine Convention, including at local scale.

The challenge we highlight for the agriculture domain relates to the need of widening income sources for alpine farmers while favouring agro-environmental practises. Indeed, by maintaining high quality agricultural productions, their social and environmental externalities (including benefits for biodiversity and multiple ES) would be sustained. One major asset of alpine agriculture is to be associated with specific ‘terroirs’ and our analysis confirms the potential for high added-value productions (as proposed by IG among others). In addition to economic incentives (e.g. PDR - MAEt), we stress the importance of extension organisations to support multifunctional farming, as mentioned repeatedly across literature and interviews.

Finally, we point out the need to increase ecological knowledge to better anticipate direct and indirect environmental and social effects of policy instruments. Indeed, multiple knowledge gaps still undermine policy ability to mitigate environmental issues (Anton et al. 2010). We proposed that a ‘rebound effect’ framework has great potential to address positive and negative untargeted consequences of governance measures. In addition, our results point out that environmental scientific evidence could be better integrated into the decision-making process. Recent academic progress could help progressing in this direction (for frameworks, see for instance Pullin et al. 2009, Dicks et al. 2014). We also acknowledge the need for an increased dialog between academics and decision-makers in this endeavour. The recent launching of the IPBES represents a major progress in this perspective and participates to creating a world-wide science-policy interface regarding biodiversity and ES (Perrings et al. 2011, Diaz et al. 2015).

AGRICULTURE – TOURISME – BIODIVERSITE

Janvier 2015

Projet CONNECT - Synthèse transversale

Quelle gouvernance pour la biodiversité et les services écosystémiques dans les Alpes ?

Quels sont les facteurs qui favorisent ou limitent l'efficacité des politiques de biodiversité?

Quelles recommandations peut-on formuler ?

BIODIVERSITE

Optimiser l'articulation des outils de la gouvernance entre échelons politiques.

But : favoriser la cohérence des politiques et rationnaliser les ressources budgétaires.

⇨ La **région** est un échelon pertinent pour la planification environnementale sur le massif des Alpes, à la condition de fonctionner en forte cohérence avec les outils du **département** qui est aujourd'hui nécessaire pour assurer la bonne mise en œuvre des politiques de conservation de la biodiversité.

⇨ Pour optimiser cette articulation, un niveau de communication approfondi, permanent, et indépendant des agendas électoraux est nécessaire entre acteurs à différentes échelles de compétence, y compris celle de la **commune** et de **l'intercommunalité**.

⇨ Une communication améliorée sur les motifs et les objectifs des politiques conçues au niveau européen est un fort atout pour leur compréhension et acceptation par les citoyens.

Quelques outils : SRCE, ENS, ouvrages favorables à la circulation des espèces, TVB, outils éducatifs, instruments de planification de l'usage des terres.

TOURISME

(Re)positionner les projets touristiques en adéquation avec les objectifs de développement durable.

But : Limiter les impacts des aménagements touristiques sur l'environnement et diversifier les activités proposées

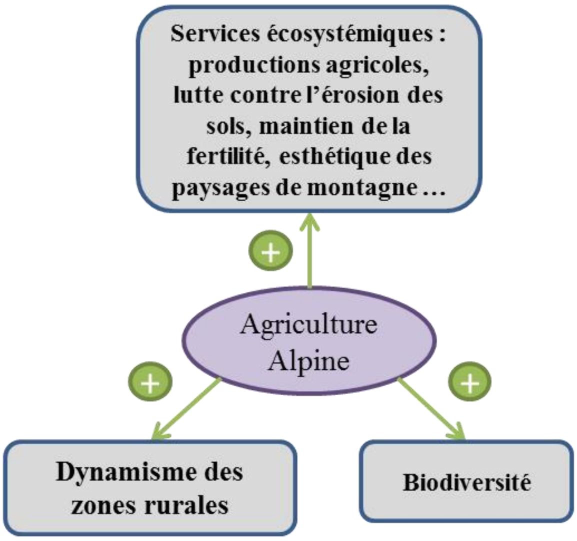
⇨ Soutenir les projets touristiques basés sur la valorisation du terroir et des espaces naturels.

⇨ Requalifier et rénover les structures d'accueil plutôt que de construire du neuf. Etendre et multiplier les transports collectifs pour accéder aux sites touristiques en montagne.

⇨ Faciliter l'accès aux décisions concernant les autorisations d'aménagements touristiques et en favoriser la transparence.

⇨ Promouvoir l'intégration des objectifs de la Convention Alpine au sein même des municipalités.

Quelques outils : Incitation fiscale, création de labels, certifications, promotion de l'agrotourisme.



RECHERCHE

Encourager les analyses conjointes des impacts environnementaux et sociaux de la gouvernance.

But : Mieux anticiper les effets indirects des politiques de biodiversité mises en œuvre.

⇨ Analyser les réactions en chaîne des politiques environnementales et les prendre en compte dans les études environnementales. Se baser sur le concept d' « **effets rebond** » qui offre un cadre novateur pour évaluer et minimiser ces impacts.

⇨ Encourager les échanges à l'interface entre sciences et action politique, en favorisant le partage de savoirs et les professions transversales.

Quelques outils : structures de R&D et d'accompagnement technique, suivi et contrôle des impacts des actions de gestion, vulgarisation des résultats obtenus.

AGRICULTURE

Elargir l'éventail des débouchés économiques pour les agriculteurs de montagne, tout en favorisant les trajectoires agro-environnementales.

But : Maintenir des productions agricoles de haute qualité en montagne et maximiser les bénéfices de l'agriculture sur l'environnement et le tissu rural.

⇨ Favoriser les modèles agroenvironnementaux d'exploitation agricole. Optimiser l'appropriation des outils politiques visant à cette conversion en faisant recours aux organismes d'assistance et de conseil aux agriculteurs.

⇨ Amplifier la dynamique positive entre renommée des produits agricoles alpins de haute qualité et promotion des espaces alpins qui attirent les visiteurs. Poursuivre la mise en place de projets valorisant une agriculture ancrée aux territoires et terroirs.

Quelques outils : indications géographiques certifiées, projets agrotouristiques, mention valorisante 'Montagne', mesures du PDR (y compris MAEt).

Projet CONNECT : http://www.connect-biodiversa.eu/
Sandra Lavorel, Directrice de recherche CNRS, LECA
Emilie Crouzat, Doctorante, LECA
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Figure 5: Policy brief proposing our recommendations for a greater integration of environmental concerns in natural, agricultural and tourism areas of the Alps.

V. Discussion

A. What can we conclude from our governance analysis?

Our objective here was to test the suitability of the method proposed by CONNECT partners for a practical implementation in our alpine case-study. This method aims at assessing the environmental effectiveness of governance (i.e. at evaluating its success) by an approach of its formal institutions (i.e. its policy instruments) and of their articulation as a policy mix. We tested two sets of criteria: usual 'CONNECT' criteria and a novel 'rebound effect' framework.

1. Conclusions on the CONNECT usual criteria

To begin with, we assessed usual criteria targeting i) policy impact, ii) fitting with the broader social context and iii) interactions within the policy mix. I stress four points for attention, two concerning the method (i.e. the criteria assessed) and two highlighting interesting trends of the alpine mix.

First, all criteria assessed following the CONNECT analysis grid **contribute to the assessment of environmental effectiveness** and spanned **a wide and interesting range of characteristics** (environmental – economic - social). **Information was generally available** to carry out the assessment and stakeholders appeared comfortable at discussing them. Thus, the cost-benefits analyses that are usually performed for assessing governance effectiveness should always be complemented by insights from socio-cultural explorations and by explicit consideration of the broader mix to look for complementarities, redundancies, conflicts and overlaps. In addition, highlighting overlapping or conflictual instruments appears highly useful to point out the **controversies representative of social debates** unresolved to date. Instruments can be developed to address the various stances legitimated in the social debate (e.g. PNAL). Such compromise and collaborative integrative tools are most required to maintain dialogue and articulate the various concerns and priorities discussed.

Second, while we informed only a **restricted definition of efficiency** (i.e. only net budgets), this criterion should be addressed in terms of marginal costs and benefits so as not to negatively weight instruments supported by important budgets without considering their related environmental gains (e.g. the economic instruments in our analysis). To date, scarcity of information on marginal effects of the instruments limits this endeavour, and I propose that **scenario-based approaches** should be used to progress in its understanding.

Third, our policy mix was overall characterised by a **high consistency and legitimacy** demonstrating a sound articulation of environmental policies with the institutional context and a rather strong cultural acceptability. Additionally, **generalised complementarities** among instruments enhance the overall environmental benefits of the alpine policy mix. Conversely, **overlaps and conflicts** undermine its effectiveness by blurring the potential usefulness of some instruments (e.g. PAEN) or by complexifying their actual implementation throughout the administrative layer cake (e.g. PTCA).

Fourth, **monitoring and control procedures** are key points to ensure credibility and actual implementation of all instruments. When they are perceived under-optimal, their instruments loose in effectiveness, at least in the mid- to long-term. I believe this holds particularly true for instruments based on **economic incentives** that require funding and for which traceability is a requirement (e.g. PHAE2), as well as for instruments relying on **consumer's preferences** as the added value of final products must be being justified explicitly (e.g. IG).

2. Conclusions on the rebound effect framework

We additionally performed a tentative assessment of rebound effects, seen as a test for new concepts developed as part of the CONNECT project. I stress three points for attention, one concluding on the framework under test and two concerning its outcomes for the study area.

First, the general framework for the assessment of rebound effects appeared to **have high potential** for warning against negative side-effects and for promoting potential synergies between biodiversity and other environmental variables. Two rebound effects appeared particularly insightful and could be informed by usually available information: **biodiversity rebound I (spatial spill-over) and service rebound**. I believe that progresses in conceptual framing and data availability are needed regarding the three other rebound effects: biodiversity rebound II (other facets of diversity), ecological rebound and environmental rebound. In particular, I see a research need for i) more precisely identifying **which facets of diversity or ecological functions should be looked at**, and ii) framing the **spatial boundaries and behavioural options that can be explored** for the environmental rebound effect. At the same time, policy design would need to specifically consider the incorporation of rebound effects in environmental assessments and in monitoring and control procedures.

Second, we found **numerous spatial spill overs** that negatively impacted biodiversity in areas not targeted by the instruments. This effect is challenging as it relates to the necessity for extended environmental assessments that would include larger spatial extents and that would rely on ecological and economic understanding of spatial dependencies.

Third, we stressed **numerous synergies that benefit to all categories of ES**, although not targeted specifically by the instruments assessed. This result supports the current move towards of joint consideration of ES and biodiversity in environmental policies (Maestre et al. 2012). We nevertheless acknowledge the need for further evidence on the links between ES and biodiversity so as to adequately design policy instruments for their joint management (Zupan et al. *submitted*).

3. Conclusions on the articulation within the policy mix

We pointed out **three clusters of instruments** that i) confirmed strong synergies among instruments related to the maintenance of alpine agriculture, ii) described a sound articulation of policies concerned by ecological connectivity at multiple scales, and iii) highlighted the challenges of governing land allocation in a constrained setting.

Articulation of instruments across scales appeared challenging and conditioned effectiveness.

Regarding the second cluster (ecological connectivity), I consider the sequence of instruments describe at decreasing scales as a **successful top-down approach** to environmental governance. Nevertheless, I believe that its good performance is highly linked to a **simultaneous bottom-up dynamic**, although not made explicit here. This dynamic would rely on the local knowledgeable and environmentally-conscious stakeholders, for instance those working in agricultural extension organisations (Grard 2010), that are able to operationalise policy measures according to on-the-ground specificities, constraints and assets (Felipe-Lucia et al. *in prep*). Additionally, this bottom-up dynamic would also rely on public participation, as “success is more likely when communities play some role in rulemaking and monitoring processes” (Epstein 2015).

Contrary to this good articulation across scales, cluster 3 (land planning) highlighted the difficulty of maintaining through scales i) **coherence of objectives** and ii) **legibility and transparency of procedures**.

The next step of our governance analysis could therefore interestingly explore the determinants of successful or problematic articulation of instruments through scales as well as the suitability of specific scales for managing certain aspects of ES supply or biodiversity conservation. Considering a higher number of policy instruments at each scale of concern (from European to municipal levels) appears necessary to be able to draw robust conclusions.

B. Three limits of this policy mix analysis

A first limit of this analysis concerns the **difficulty to assess thoroughly the performance** of individual instruments. Indeed, as mentioned in the previous sections, some criteria were quite **subjective** (e.g. legitimacy, consistency) while for others, we relied on **partial information** to conclude (e.g. no information on marginal costs and benefits to assess efficiency). Although we did our best to expand the sources of information by consulting both expert and academic literature, and additionally consulted experimented stakeholders, our analysis could not be exhaustive. More robust results could be proposed through **additional stakeholder consultation**, either individually or also as collaborative working groups confronting various opinions and concerns. Additionally, **methodological progresses** are still required regarding the rebound effect framework that, although promising, remained challenging to inform.

A second limit relates to the **challenge of expanding our focused results** at the scale of the broader policy mix. Indeed, our approach was focused on i) a restricted set of ES, ii) selected interactions among them and iii) specific instruments currently used for their management. Integrating further complexity in our analysis regarding these three steps of focus would enable getting a broader perspective on the alpine system assessed, in particular by considering **extended ES bundles**. As a consequence, I propose this policy mix assessment as an **entry point for discussion**, as a basis for increasingly considering environmental issues in management. In other words, our analysis should not be understood as a normative judgment on the current alpine governance system, but rather as an opportunity to discuss some of its interests and potential pitfalls.

A third important limit to expanding our results is that we focused on **formal institutions only** and therefore did not assess the importance and effectiveness of **informal institutions** (networks, values, norms, traditions and beliefs). However, informal modes of governance, and in particular networks including state and no-state actors, have attracted much attention over recent years. In combination with the increasing reference to market-based instruments, they seem to gain importance relatively to formal regulative institutions (Pahl-Wostl 2009, Lascoumes & Simard 2011). As a consequence, a comprehensive vision of alpine governance would require considering a diversity of institution types, even though I acknowledge the challenge represented by the assessment of environmental effectiveness of less formal institutions. In the context of global changes, such vision would also inform on the potential **resilience** of the system (Folke 2006). Resilience has been described as relying on a diversity of governance mechanisms combining strength and sustainability of commitments from a central power (e.g. regulatory instruments) with flexibility and social participation inspired by adaptive governance (e.g. informal networks, voluntary instruments…) (Undertal 2010). Our analysis is not comprehensive enough to assess the resilience of alpine environmental governance. Using inclusive participative frameworks could be an interesting starting point to assess the various formal and informal institutions associated with the governance of ES and biodiversity. Among those, I suggest in particular the mental model mapping (e.g. Moreno et al. 2014) that accounts for multiple viewpoints, sources of knowledge and factors analysed in a straightforward and practical way.

C. Social determinants and impacts of policy mixes

Constructing a policy addresses much broader determinants than “seeking the "best" or most cost effective "solution"” to a given problem (Simeon 1976). Some authors even support that “instruments are rarely selected on the basis of their implementability and effectiveness” (Bressers & O’Toole 1998, in Lascoumes & Simard 2011).

Rather, policy mixes would be constructed following **their social acceptability, the habits of specific policy fields and the political constraints faced by decision-makers** (Lascoumes & Simard 2011). For instance, our analysis showed that participative processes overall reinforced the legitimacy of the instruments and were judged positive for enhancing effectiveness (e.g. SRCE). Our results are in accordance with current governance trends that consider participative instruments as constitutive elements of policy mixes (Pahl-Wostl 2009, Young 2011). However, up until 30 years ago, participative processes had mostly a symbolic dimension (i.e. a rhetoric effect). Their implementation was challenged by the fact that they did not ‘*fit’* well in the ‘command and control’ traditional governance model (Lascoumes & Simard 2011). And yet, nowadays, participative processes have become iconic instruments of the current ‘new governance’ paradigm, to the point of being called “a new tyranny” by some authors (Cooke & Kothari 2001). Acceptability and use of individual policy instruments is therefore variable in time and representative of the socio-cultural context of their implementation (Pahl-Wostl 2009). It can be noted that such contextual dependence of the policy mix acceptability rather promotes stasis and hinders transformation than supports adaptive capacity.

As such, analysing the ***how*** of environmental governance, as we have done, should be complemented by analysing the ***what*** (i.e. the scope - the aspects considered by decision-making, and those that are not) and the ***who gets what*** (i.e. the distributive dimension of costs and benefits among the members of the society) (Simeon 1976). These two aspects (scope and distributive dimension) were beyond the reach of our study but I stress the necessity of looking further than actual results of implemented policies to understand their social determinants and consequences. For instance, through the rebound effect analysis, we pointed out some untargeted effects of policy instruments. This could be of help to identify the variables ‘out of policy scope’, for instance spatially (e.g. deprived valleys of low tourism and agricultural reputation), but also ecologically and socially. In the same line, to progress regarding distributive dimensions, we could further investigate the ecological and social impacts of the generalised economic support of alpine farmers (e.g. through CAP subsidies) on the resilience and adaptive capacity of the agrosystem as well as on social equity. Approaches based on **scenarios** proposing alternative economic incentives could be interestingly explored (e.g. Palomo et al. 2011, Nettier et al. 2012, Lamarque et al. 2013).

D. Governing change – an advocacy for social learning

Integrating social learning into governance processes can be proposed to make them adaptable and able to accompany stakeholders in addressing complex and dynamic management issues (Armitage et al. 2008). We did not address **learning capacities** and **evolutionary potentials** of instruments and of their articulation in our assessment. Nevertheless, environmental effectiveness could be achieved through iterative learning cycles shaping progressively the instruments to their paradigm and their objective. A **multi-loop concept for learning** has been developed to “take into account the different levels that provide guidance and stability in a social system at increasing time scales for change” (Pahl-Wostl 2009). As defined by

Armitage et al (2008), *single-loop learning* will refine actions to improve performances, by identifying "alternative strategies and actions (e.g. harvesting techniques) to resolve specific problems and improve certain outcomes (e.g. improved incomes, higher yields)". Policies can be adapted by changing thresholds of reference (e.g. the minimum size of tourism infrastructure considered for authorisation procedures as UTN). In turn, *double-loop learning* will question guiding assumptions and change the frame of reference, "resulting in fundamental changes in stakeholder behaviour". Policies can be adjusted to fit with this new frame (e.g. adaptation to climate change can be sought by restoration of floodplains, and not only through an increase in the height of dikes (Pahl-Wostl 2009)). Finally, *triple-loop learning* refers to a transformation of the structural context and can imply a change in paradigm. New governance norms and protocols are proposed. I hypothesise that the introduction of ES as targets within land planning and conservation policies may initiate triple-loop learning. Indeed, frameworks have been proposed to support ES adaptive management (e.g. Daily et al. 2008) and learning processes have been proved to affect the management of ES in scenario-based approaches (Lamarque et al. 2014). Among the characteristics proposed by C. Pahl-Wostl (2009) to identify changes in governance regimes expected after triple-loop learning, the ES concept already induced changes in conservation policies at various scales, it acknowledged uncertainty and opened the way for considering different perspectives in decision-making. Moreover, a new category of services, 'the adaptation services', has been recently proposed regarding climate change (Lavorel et al. 2015). They are defined as "the benefits to people from increased social ability to respond to change, provided by the capacity of ecosystems to moderate and adapt to climate change and variability". Managing these services, regarding climate change but also other global changes, will require new approaches and adapted regulation frameworks. Whether the concept of ES will finally deliver a 'triple-loop effect' will be only assessable later on, as time is needed to ascertain changes in stakeholder networks, related power asymmetries and actual inclusion of environmental issues in governance.

VI. Synthesis

Our approach of governance of ES and biodiversity in the French Alps led us to focus on a set of 10 formal instruments managing influence relationships at the interfaces between agriculture, tourism and biodiversity. Through a fruitful collaboration with Elise Trouvé-Buisson, the Master student in charge of this assessment, we performed an extensive literature review to assess the environmental effectiveness of these instruments and further investigated their positive and negative rebound effects. Our approach was supported by consultation of regional experts that validated and complemented our findings, although more robust findings could still be proposed after extended stakeholder consultations. Successive synthesis steps were undergone until we obtained a final assessment of the performance of each instrument according to a set of 13 criteria. We paid particular attention to the articulation of each instrument within the broader policy mix and explored their mutual interactions.

The whole approach can be considered as a practical pilot implementation of a methodology proposed by our CONNECT partners at a more theoretical level. The method we followed is firstly based on a set of usual criteria targeting i) policy impact, ii) fitting with the broader social context and iii) interactions within the policy mix. This information was complemented by a novel rebound effect framework that dealt with untargeted positive and negative environmental impacts. From our experience, information was overall available to assess the set of usual criteria even though some of them remain quite subjective (e.g. legitimacy) and other lacked precise information to be comprehensively assessed (e.g. efficiency). I believe that an extended stakeholder consultation would be adequate to strengthen our results. Regarding the rebound effect framework, our experience supported its high potential for a more sustainable environmental management. However, progresses are still required at a conceptual level to propose practical definitions that would be more easily understood and informed. Increased attention to these rebound effects in academic studies and real policy mixes appears an interesting way of progress through adequate monitoring and control procedures.

As results, we answered our first research question, relative to the characteristics of individual instruments and to their environmental performance. In particular, we pointed out the overall high consistency and legitimacy of the instruments assessed, demonstrating a sound articulation of environmental policies with the institutional context and a rather strong cultural acceptability. We stressed the importance of adequate monitoring and control procedures to ensure credibility and actual implementation of all instruments. Their careful design is necessary to account for the numerous potential spill overs we detected in the rebound effect analysis. Finally, we stressed the ability of some policy instruments to synergistically sustain ES and benefit to biodiversity. Nevertheless the rebound effect analysis detected numerous potential spill overs, highlighting the need for better knowledge and communication on influences among social and environmental drivers, biodiversity and ES.

Our second research question regarded the articulation of instruments within the broader policy mix. We found that the generalised complementarities among instruments enhanced the overall environmental benefits of the alpine policy mix. In some cases overlapping domains and scales of application appeared as a barrier to implementation. Conflicts were rarer. We produced three synthetic messages on: i) the synergistic support of alpine agriculture by multiple instruments, ii) the good performance of instruments at nested scales to enhance ecological connectivity, and iii) the challenge of interactions among land allocation instruments.

Finally, we explored our third research question relative to the potentials of policy mix analysis to inform management of ES and biodiversity. So as to communicate on the findings previously presented, we produced a policy brief targeting stakeholders of intermediate level (i.e. regional mostly). Further I discussed the additional aspects that would enrich our analysis so as to be able to accompany stakeholders effectively in environmental management, including the need for social learning and for more integrative consideration of informal institutions.

A synthesis of our governance analysis is proposed in Figure 6.

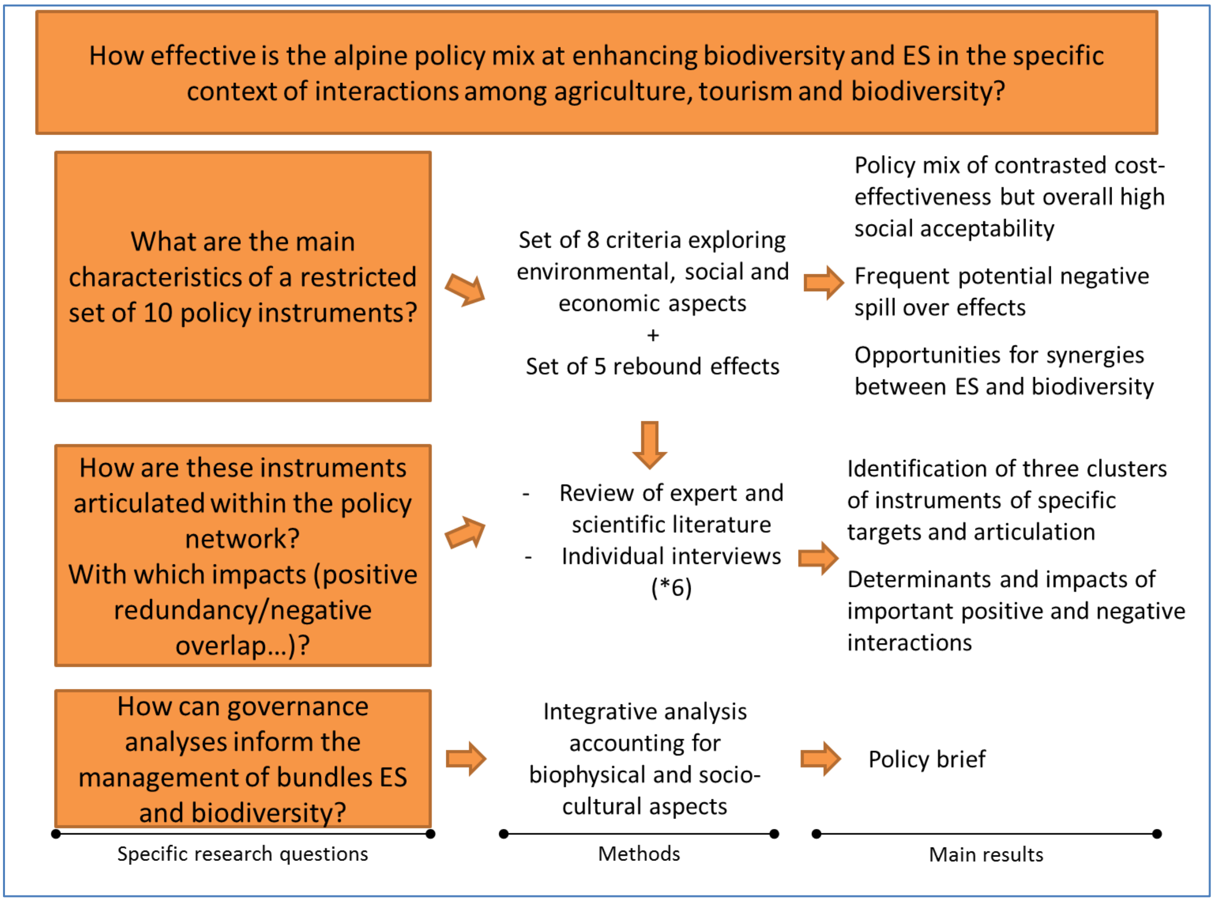

Figure 6: Specific research questions explored in the policy mix analysis of instruments managing influences between agriculture, tourism and biodiversity (Chapter III), related methods and main results obtained.

Although this is definitely only a first approach due to lack of disciplinary background and time, I found this policy mix analysis both challenging and necessary. Challenges were principally to enter a highly complex mix, to integrate multiple scales and concerns, and to mix insight of political sciences with our environmental perspective. Necessity referred to connecting ecological and social findings with the tools that actually frame natural resource management. I therefore consider this governance exploration as a bridge that enables a more comprehensive dialog with stakeholders. Indeed, their management of natural resources is the result of numerous compromises that include biophysical constraints, stakeholder interplay and political outcomes partly revealed by policy instruments.

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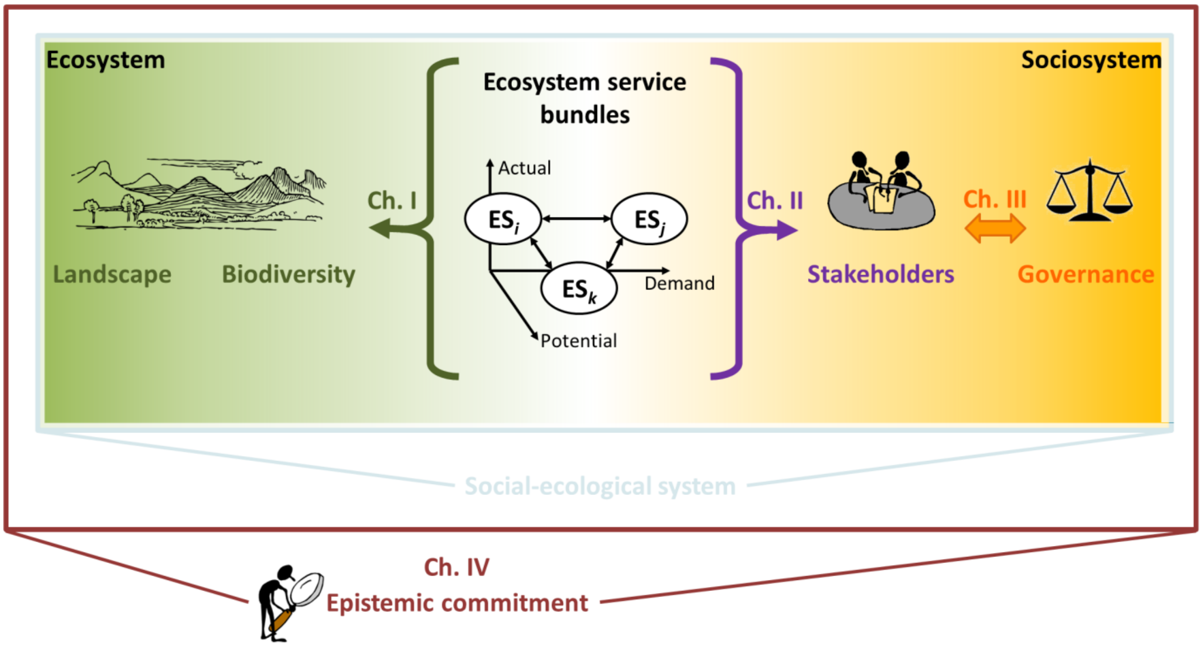
Chapter IV - Exploration on epistemic commitment in ES research



Chapter IV - Exploration on epistemic commitment in ES research

Chapter IV aims at taking a step back on the assessment of networks of ecological parameters (i.e. ES and biodiversity) through the exploration of some conceptual and ethical issues linked to this research domain. It addresses the interrogations I faced while discovering the ES concept and related controversies, as well as the questions I sought to answer regarding roles of scientists in society. This research was not part of the objectives of my PhD *per se*, it rather responds to my personal questioning that took form all along the three last years. As such, this chapter does not claim exhaustiveness in concepts grasped, literature browsed or controversies explored. I do not pretend having an in-depth and critical understanding of pros and cons in epistemology, axiology and philosophy of sciences in relation with ES. This chapter rather exposes the questions I faced during my PhD and proposes some solutions and thinking I came across in literature and that I often discussed with scientific partners in my team. Overall, I conceive Chapter IV as a practical essay on concepts, values and commitments that I found necessary to better understand so as to progress in ES science.

Chapter IV is structured in five sections:

- Section I presents the **specific research questions** related to our conceptual and ethical exploration of issues raised by research on ES.
- Section II exposes the general setting linking **science, governance and ethics** within which ES research is developed and focuses on determinants and consequences of **uncertainty, value pluralism and controversies**.
- Section III characterises distinct **epistemic commitments** embodied by ES researchers and explores their **personal and social consequences**.
- Section IV links **the work I contributed to** during my PhD and related projects to the **academic postures** described, and discusses issues of interfaces between science, governance and society.
- Section V **concludes** on my personal though process regarding controversies, values and scientific commitments in ES science.

I. Specific research questions

The overarching objective of this chapter is to explore the various epistemic commitments (i.e. scientific postures) related to academic research on environmental topics, their determinants as well as their personal and social consequences. This objective was approached through the three following questions:

1) What is the epistemic and social context leading researchers to be (potentially) at the interface between science and governance?
2) Which typical epistemic commitments are usually described in biodiversity and ecosystem service research domains? What are the consequences at individual and social levels of these commitments?

3) How does the work carried out during my PhD match with typical epistemic commitments? What are the consequences of being conscious of these conceptual and ethical issues?

To answer these questions, I explored an interdisciplinary literature from ecological, economical and philosophical backgrounds and aimed at interweaving their insights to characterise, in the current academic and social setting, the postures adopted by environmental scientists in general and in my work in particular.

II. When environmental sciences interface with governance and ethics

A. Setting the stage for relationships between science and society

Science has been traditionally described, and claimed, as disconnected from the turpitudes of governance, so as to maintain (a supposed) objectivity in knowledge production (Pielke, 2007). However, the real science – policy interface can be hypothesised as much blurrier, due to renewed demands from society upon scientists in the context of global change, to the rise of science in social participation (and vice-versa), and to individual scientists themselves demonstrating the will to engage with policy. In short, "if scientists ever had the choice to remain above the fray, they no longer have this luxury" (Pielke, 2007).

1. How can society make use of science?

Two models have been proposed to describe the modalities under which knowledge acquired by science can contribute to decision-making.

Since the end of the Second World War, links between science and policy were understood as a continuum, known as the *linear model* (or reservoir model), where knowledge follows a directional flux from basic research to applied research. Basic research is disconnected from any application, pure and general as described in Weinberg's axiology of science (Weinberg, 1970); it is focused "more on the creation of knowledge, as an end in itself" (Pade-Khene, 2013). Picking up on this pool of knowledge, applied research is intended to address 'real-world' problems, with the objective of contributing not only to their understanding but also to their solving. Overall, this linear model suggests that "achieving agreement on scientific knowledge is a prerequisite for a political consensus to be reached and then policy action to occur" (Pielke, 2007).

An alternative model has been proposed through the *stakeholder model* (Pielke, 2007). Knowledge is conceptualised as depending on complex feedbacks between researchers and users of science, the latter gaining a role in knowledge production while the former are partly responsible for uses of science in decision-making. This model acknowledges that policy-relevant science is not value-free and that consequently knowledge production should be discussed by scientists and science-users as a shared responsibility. It is the basis for *citizen science*, understood as a science that acknowledges that fact that researchers are both scientists and citizens (Coutellec, 2012a). Here, citizen science does not refer to the participation of citizens in the practical scientific process (e.g. through voluntary measures of environmental variables). Rather, citizen science highlights the possibility for a scientist to base his/her citizen choices partly from his/her scientific knowledge, and to partly drive his/her research from its values and concerns.

The IPBES (Intergovernmental Platform on Biodiversity and Ecosystem Services) is intended to work according the stakeholder model, as "it is expected that these stakeholders will act both as contributors and end users of the platform" (http://ipbes.net/stakeholders.html) (stakeholders considered by IPBES are governments, United Nations organisations, Multilateral Environmental Agreements, stakeholders from the scientific community and broader civil society, including non-governmental organizations and the private sector). Although inspired by the IPCC (International Panel on Climate Change https://www.ipcc.ch), IPBES endeavour differs from its posture (Brooks et al. 2014). Indeed, for IPCC, science has to be "policy-relevant and yet policy-neutral, never policy-prescriptive", following a more classical linear model. This difference regarding the vision of science in IPBES constitution could answer the claims that the interpretation of scientific results for policy concerns can hardly be thought as neutral while facing uncertain and controversial issues as climate change adaptation, biodiversity loss or environmental management (Pielke, 2007). Once scientific evidence has been communicated to the non-scientific community to inform a given problem, interpretation of its significance for alternative policy options remains indeed challenging.

2. How can scientists contribute to democracy?

In his book *The Honest Broker: Making sense of science in policy and politics* (Pielke, 2007), Roger Pielke proposed two visions of democracy useful for understanding how individual scientists can support decision-making.

Democracy has been perceived as a pluralism of groups of interests that get opposed in political debates. In this case, scientists willing to contribute align with the group supporting their opinion. They offer their expertise and legitimacy that can be seized as arguments in favour of a given point of view. This vision has been described as a *Madisonian democracy*, after the writings of the political theorist and President of the United States James Madison at the end of the XVIIIth century. Beyond an opportunistic use of science (i.e. only when it fits someone's storyline), this vision promotes the use of science "with purpose": it supports scientists to advocate in proactive manners for their favourite option during political debates

Alternatively, democracy has been described as a competitive system by the political scientist E. E. Schattschneider in 1975. Under this conception, elites are in charge of determining a set of options given as relevant to face a specific issue. Public is called to participate by expressing its preferences between this set of 'expert-approved' options as the next step of the political process. Scientists help policy makers and the public by clarifying the implications of actions proposed on the basis of their scientific knowledge, without taking side.

These two conceptions of democracy fundamentally differ regarding the position of the *expert*, i.e. the one with the ability to provide policy significance to scientific results. In a Madisonian conception, scientists can be part of the political debate and of the decision-making process, and are even encouraged to do so. At the opposite, democracy as conceived by Schattschneider strictly maintains its experts external to the governance process and diffuses the idea of a neutral science. As an example, the French procedure for ecosystem assessment (EFESE – the French implementation of MEA) was designed following a Schattschneider vision of democracy. Indeed, its first objective is to provide a biophysical and socio-economical assessment of ecosystems and ES at national scale. Then, EFESE should use scenarios to assess the alternative futures of ES under the main general policy options currently discussed. Therefore, EFESE scientists will clarify the implications of different governance choices without using its expertise to support one upon the others.

Overall, these differing conceptions of democracy induce the need of being explicit on how to interpret guidance received from an expert. Indeed, society in general and decision-makers in particular should know whether the knowledge that is proposed as a support for decision seeks i) to advocate for a particular policy option or ethical setting or rather ii) to deliver information, as an outsider, on expected consequences of governance choices.

3. How can science become more democratic? Conditions for an epistemic democracy

The previous sections conveyed the idea that axiology of science, i.e. values shaping and characterising scientific work, needs to be questioned explicitly to progress toward a constructive dialog with society (Weinberg, 1970). If science is not a neutral monolith disconnected from governance, scientists can turn into *citizen scientists*, i.e. "people who intertwine their work and their citizenship" (Stilgoe, 2009). The collaboration between citizen scientists and also between such scientists and the broader society creates the opportunity for an *epistemic democracy* (Coutellec, 2012a), understood as the production of multiple strands of knowledge contributing to sound interactions with society i.e. a citizen and socially relevant science concerned by decision-making. Epistemic democracy is conditional to a true *transdisciplinary* approach of sciences, an approach where various disciplines collaborate to produce empirical and pragmatic knowledge while also becoming "a social process dealing with values and norms of both society and science" (Reyers et al. 2010).

Three conditions have been proposed as a basis for this epistemic democracy (interested readers are referred respectively to their thorough description in the following papers: Coutellec, 2012a, b, c).

Firstly, the epistemological condition of such science relies on **scientific pluralism**. This pluralism concerns disciplines, styles of scientific reasoning and methods employed. In reality, one form of knowledge is often preferred upon the others due to an easier communicability or a stronger social recognition (e.g. statistics from hard science vs narratives from soft science) (Jax et al. 2013). Moreover, this preference promotes without explicit questioning particular methods and units for the assessment of the issue addressed at the expense of others. Such situation of 'epistemological silos' (Miller, 2008) threatens the democracy of science while only the acknowledgement and consideration of multiple epistemological logics would lead to a co-constructed and legitimate understanding of a complex issue (Stilgoe, 2009, Coutellec, 2012a). In ES science, assessments mobilising different sciences (i.e. ecological, social and economic sciences) have been found to score ES differently (Martin-Lopez et al. 2014). This finding is to be related with a widespread tendency to neglect socio-cultural dimensions, mostly compared to economic valuation (Chan et al. 2012), thereby promoting the language of hard sciences (quantitative assessments leading to statistical analyses in biophysical or economic terms) over the language of social sciences (often deriving from narratives or qualitative data). In the same idea, the demand facet of ES is still under-assessed compared to supply, although frameworks have been recently proposed to bridge this gap (for instance Bastian et al. 2012, Burkhard et al. 2012, Villamagna et al. 2013, Crouzat et al. in prep). A general effort in ES science should therefore bring ecologists, political economic and social scientists to increasingly work jointly towards multi-dimensional and multi-disciplinary assessments.

Secondly, the ethical condition for an epistemic democracy is based on **axiological pluralism**, i.e. the recognition of multiple values as joint objectives to knowledge production. However, the idea that science is value-free has been long defended and is to be linked to positivist logics. For instance, in the XVIIth century Galileo warned that "the facts of Nature […] remain deaf and inexorable to our wishes". Early in the XXth century, the mathematician Poincaré still proposed a similar vision of ethics and science, which "never conflict as they never meet". However, as extensively discussed by Professor of philosophy Hugh Lacey "science may be appraised, not only for the cognitive value of its theoretical products, but also for its contribution to social justice and human well-being" (Lacey, 2002). This opinion challenges the idea of a disconnection between facts and values. Ethics of science usefully proposes a framework to control research procedures and assess its productions. Peer review procedures, detailed publication of methods and results, as well as declared absence of conflict of interest are given as the basis for this scientific integrity, leading to a (supposedly) shared 'deontology' within the scientific community. However, ethics of science is linked to epistemic responsibility and therefore does not necessarily help scientists facing the ethical responsibility with which they engage through their work (Coutellec, 2012b). Therefore, A. Coutellec calls for a *generic ethics* in order to add ingredients from multiple ethical thinking to the research process in a cumulative and non-substitutive way, so as not to forget humans in science. In ES research, value pluralism seems to be the rule, be it among individuals (Sandbrook et al. 2010, Hermelingmeier 2014) or disciplines (Maitre d'Hôtel & Pelegrin 2012, Arpin & Granjou *in press*) (but see also the dedicated section below for details). As an example, Jax et al. (2013) proposed that four types of values could be attributed to non-human nature: *inherent moral value* (also called *intrinsic* value, i.e. "deserving direct moral consideration for their own sake"), *instrumental value* (i.e. in principle "replaceable, compensable and (in the extreme) [that] can be price-tagged"), *fundamental value* (i.e. related to "the most basic, systemic and complex conditions for existence"), and *eudaimonistic value* (i.e. necessary for "a life worth of a human being"). Those four types of values can equally be seized as arguments in favour of biodiversity conservation or ES supply maintenance, but are linked to the very distinct value-backgrounds embodied and should therefore be considered in their diversity in ES research.

Thirdly, the anthropological condition for epistemic democracy lies in the recognition of a **temporal diversity** of sciences (i.e. *chronodiversity*). A Slow Science movement (http://slow-science.org/), in analogy to the Slow Food movement, has begun spreading in Europe since 2011. As described by A. Coutellec (2012c), this movement calls against the widespread culture of immediacy that puts under pressure individual scientists and threatens the quality of science. Its objective would be to get out of the obsession of scientific productivism (publishing for publishing). At the opposite, a slow science would support new places for science production where long term would be preferred upon short term and where time would be given for appropriating knowledge. Such science would permit progressing toward knowledge of quality for the general interest, in complement to sciences driven by other rhythms. ES sciences have known a very active development in the last 25 years, expressed by the exponential increase of publications based on the concept since the late 1990s (Dick et al. 2011). This very rapid rise led to a temporal overlap between on the one hand definition and stabilisation of concepts and methods, and on the other their practical implementation and use by decision-makers and managers (see Barnaud & Antona, 2014). The global dynamics of the scientific sphere could therefore expose ES research to the pitfalls of a Fast Science. Nonetheless, temporal diversity could be approached through a broader perspective that would consider the dynamics of research teams and individuals. Indeed, the progressive articulation of research projects dealing with ES within a scientific team enables capitalising on what has been achieved over years. In short, even though ES research induces working with a 'hot concept', chronodiversity could be reached through the combination of projects, publications and research networks in which teams engage.

Overall, in ES research as in others, recognition of scientific pluralism, renewed relationships between science and ethics and scientific chronodiversity would give researchers the opportunity to build knowledge characterised by value pluralism, engagement and co-construction with society. As such, research becomes a 'civic act' (Coutellec, 2012c).

B. Specificities of environmental and ES sciences

Environmental sciences in general, and research on ES in particular, articulate the need to maintain functioning ecosystems with a sustained human well-being (Jax et al. 2013). By establishing a "bijective relationship between ecosystems and societies" (Barnaud & Antona 2014), science focused on the management of environmental resources faces some additional issues linked to the implication of stakeholders, uncertainties specific to knowledge 'in the making' and controversies linked to the ES concept.

1. An increased call for participative sciences

Since the 1960s, civil society has increasingly voiced its concerns and opinions regarding governance of complex problems, including management of biodiversity and environmental resources (Pade-Khene et al. 2013). *Politics*, understood as the process of negotiation and compromise that precedes decision from policy-makers (Pielke, 2007), had since then accounted for multiple groups of interests, even though the degree of their inclusion remained highly variable (Arnstein, 1969). Thus, in environmental management, **participatory planning** and **co-management** of resources have become widespread at least in discourse but also increasingly in practise (Menzel & Buchecker, 2013). A *stakeholder* is "any group or individual who can affect or is affected by the ecosystem's services" (after Hein et al. 2006 in Hauck et al. 2014). They need to be identified and involved as a result of the widespread dynamics towards the inclusion of civil society in governance. Their participation is seen as a mean to "influence and share control over development initiatives and the decision and resources which affect them" (World Bank, 1996 in Luyet et al. 2012).

Much academic progress has been made for a better identification and inclusion of relevant stakeholders (see for instance Reed et al. 2009) and expected outcomes for their involvement include better trust in decision, improved project design and management and fostering of social learning (Luyet et al. 2012). However, experiences of research engaged in stakeholder participation processes often highlight the complexity of interacting with these groups of multiple and potentially differing opinions, values and backgrounds (Pade-Khene et al. 2013). Moreover, scientists involved in participative research face an expensive and time-consuming process, that can further induce frustration for stakeholders that would not fully supports its implementation (or for scientists that would not support the research results), or alternatively exacerbate power inequities between groups (Luyet et al. 2012).

Overall, despite these difficulties and the fact that real outcomes of participative processes might not reach expectations, they are often described as the "only way to realize the [planning] projects" (Menzel & Buchecker, 2013). Additionally, scientists in environmental sciences can hardly avoid getting engaged in these collective adventures because "as a mission-orientated, pragmatic discipline", scientists in ES research should become "involved in the messy process of collaborating with and empowering stakeholders in strategy development and implementation" (Cowling et al. 2008).

2. A crisis discipline 'in the making' that relies on irreducible uncertainties

One specificity of environmental sciences, and of ES research in particular, is linked to the **sense of urgency** in response to pressing needs such as avoiding increased biodiversity losses or ecosystem damages (Blandin, 2009). Indeed, if not dealt with early enough, it is risky to assume that species populations and dynamics could be sustainably maintained or that ecological functioning of (semi-)natural areas would be recovered. Behind these issues, there is a serious threat for human well-being both at global and local scales (MEA, 2005). Science addressing such 'hot' issues is called a *crisis discipline* or a *mission-driven discipline* (Sandbrook et al. 2011), and can ask its experts for recommendations even though the knowledge they rely on remains controversial. Indeed, "a conservation biologist may have to make decisions or recommendations about design and management before he or she is completely comfortable with the theoretical and empirical bases of the analyses" (Soulé, 1985). As such, a discrepancy is to be faced between decision-makers consulting what they conceive as a '*ready-made science'* and researchers engaged in a '*science in the making'* (Barnaud & Antona, 2014).

The main consequence of relying on such a science is that knowledge presents high **uncertainties**, and this particularly applies to research in the ES domain. The concept in itself is not yet stabilized (Barnaud & Antona, 2014) but efforts are made to reach consensual definitions (for instance, Fisher et al. 2009, Lamarque et al. 2011). Moreover, increased scientific insights should strengthen our ability to predict ecosystem responses to change. Yet, attention to ES modelling outputs seldom targets uncertainty (Seppelt et al. 2011) and accepting uncertainty might be necessary as it presents some fundamentally irreducible traits (Pielke, 2007). Indeed, social-ecological systems are complex, they include multiple, interacting and dynamic processes that lead to a widespread unpredictability of their dynamics, characterised by thresholds effects and potential tipping points altering their functioning (Barnaud & Antona 2014). This complexity generates 'myopia', that can be understood as an epistemic uncertainty that blinds stakeholders having to take decisions affected by uncaptured and yet influential global dynamics (Pielke, 2007). Yet, a recent movement towards an explicit accounting of uncertainty is spreading in the common ES-research culture (see for instance Schulp et al. 2014). Indeed, following IPCC and MEA methodologies, national and European boards responsible for ES assessments (i.e. UK NEA and IPBES) have proposed two methods to characterise uncertainty (Figure 1). For each feature assessed, "estimates of certainty are derived from the collective judgement of authors, observational evidence, modelling results and/or theory examined for this assessment" (UK NEA, 2011). These estimates are then communicated qualitatively through the four-box model, combining high/low levels of agreement with significant/limited levels of evidence (Fig. 1.A), or quantitatively through the likelihood scale, based on probability of occurrence (Fig.1.B). The two methods help addressing the need of action in a context of uncertainty, as had been previously acknowledged regarding climate change for which "lack of full scientific certainty should not be used as a reason for postponing such measures [to limit greenhouse gas emissions]" (UNFCC article 3.3 in Brooks et al. 2014). This methodological proposition can be considered as a promising step toward the generalisation of uncertainty assessment in ES science, which needs to be strengthened so as to become both scientifically accessible and culturally evident.

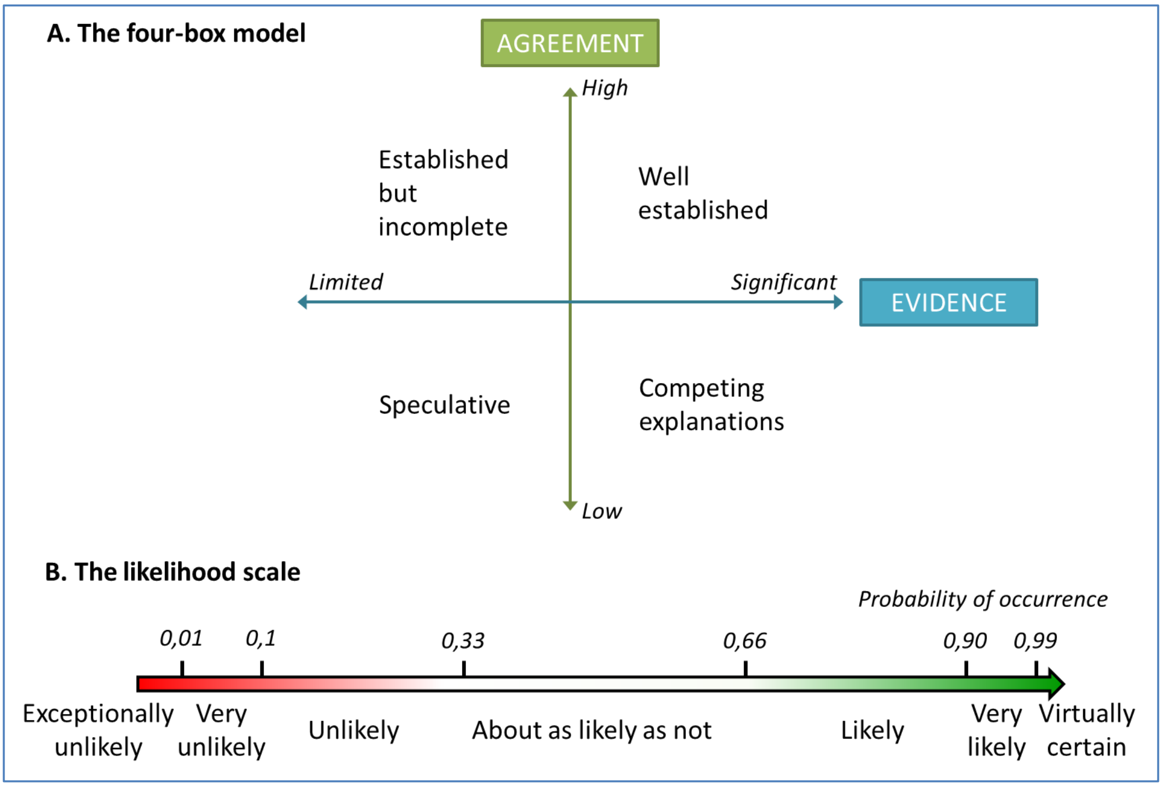


Figure 1: Methods used to characterise uncertainty in UK NEA and proposed for IPBES, initially inspired by IPCC.

In conclusion, scientists - and more generally all stakeholders concerned by environmental management - are led to cope with the fuzziness steaming from ongoing epistemic uncertainties and conceptual controversies. Thus, making definitions, possible outcomes and uncertainty levels explicit should be considered a necessity to make all concerned stakeholders aware of epistemic limitations and of social implications regarding the concepts mobilised, and thus to favour justice and equity (Jax et al. 2013, Barnaud & Antona 2014).

3. ES - A prismatic vision for human-nature relationship and related values

The very simple and broad definition of ES as "benefits people obtain from ecosystems" (MEA 2005) has seldom been criticised for itself and concentrated social debates on practical and implementation aspects of the concept. Indeed, the swift adoption of the ES language by policy-makers and stakeholders from civil society led to the *reification* of the concept, i.e. its transformation into a "concrete, tangible and measurable" object which could be assessed and managed (Barnaud & Antona 2014). However, this definition drives the way people unconsciously conceive nature and induces a particular metaphor to describe the **relationship between human and their environment** that is, at least, not universal (Raymond et al. 2013). ES exist only if someone, i.e. a human being, benefits from them and are thus framed in an *anthropocentric* conception of the world (Luck et al. 2012, Fisher & Brown 2014). Following Descola's words, nature conceived through the prismatic vision of ES is thereby segregated from culture, which favours the prominence of utilitarian values over others (Jax et al. 2013, Maris 2014). This dis-embedment of social systems from ecological systems is typical from western modern societies (Gómez-Baggethun & Ruiz-Pérez 2011) and could explain why "people have forgotten that their survival depends on ecosystems that have limited and non-substitutable resources" (Barnaud & Antona 2014). Several authors advocate for the consideration of multiple metaphors and alternative relationships to nature (recently, Binder et al. 2013, Jax et al. 2013, Raymond et al. 2013).

Difficulties arise as "in researching a social ecological system, not only the system behaviour [is] complex, but the values and the goals that society holds for that system vary as well" (Miller et al., 2008). Thus, environmental management faces **axiological pluralism** among stakeholders. Authors have sought to unravel the rationales and modalities of nature conservation or of use of the ES concept, in order to progress in the understanding of values held by scientists, citizens and decision-makers. For instance, Sandbrook et al. (2011) used the Q methodology as a mean to examine junior professional subjectivity regarding conservation values. Their interest was to characterise various viewpoints as a way to explore and help addressing the tensions over the practise of 'hot' science. The European research project OPERAs (www.operas-project.eu) similarly investigated the various perspectives of researchers involved in the project regarding the ES concept (Hermelingmeier 2014). The objective of making axiological diversity explicit was to overcome the barriers to a practical implementation of the ES concept, so as to handle it efficiently within the scientific process. These two examples, proposed among others (see for instance Arpin & Granjou, *in press*), conclude on the necessity to explicitly account for diverse ethical stances to build "honest and ultimately effective relationships" with society and accordingly shape adequate governance options (Sandbrook et al. 2011).

4. Pros and cons regarding the ES concept

The ES concept has numerous **interests**, including the potential to increase environmental concerns in land planning and management to sustain biodiversity and human well-being (MEA 2005, Vihervaara et al. 2010, Reyers et al. 2012). ES have been called *boundary objects* (Barnaud & Antona 2014) as they can be handled by various stakeholders, create dialogue opportunities among them and speak a common language that is not strictly the one of ecologists in order to support nature conservation (TEEB 2010, Costanza et al. 2014, Abson & Termansen 2014). As such, ES are considered as representatives of a new paradigm in science, understood under a Kuhnian perspective, i.e. a perspective where knowledge progresses through abrupt transformations called science revolutions (Plant & Ryan 2014).

However, what was initially conceived as an "eye-opening metaphor" (Norgaard 2010) gathered numerous **oppositions** linked to the operational implementation of ES in policy and management. Indeed, ES have been criticised because of the oversimplification they conveyed regarding the dynamic natural systems under assessment: there is a risk that ES would act as complexity blinders that do not push forward renewed global institutions and resource allocation required to reach environmental sustainability (Norgaard 2010). Moreover, many controversies are linked to the economic valuation of ES that currently dominates environmental assessments. A huge body of literature explores the process and negative consequences of this economic focus that opens the door to commodification of nature (interested readers are recommended to read, among others, Gómez-Baggethun & Ruiz-Pérez 2011, Maris 2012, Méral 2012, Jax et al. 2013, Barnaud & Antona 2014, Boeraeve et al. 2014, Maris 2014). Guidance on whether or not to perform economic valuation can be found (for instance Kallis et al. 2013), as well as comparisons between multiple languages of valuation (Martín-López et al. 2014). Additionally, ethical concerns on induced inequities among stakeholders and on the core focus on a utilitarian logic in nature conservation also raise many controversies (Luck et al. 2012). Schröter et al. (2014) recently summarized main critics and proposed counter-arguments as a "step toward an informed and structures dialogue between opponents and proponents of the concept".

Overall, the ES concept cannot be considered only as rhetorical because of associated shifts in funding, partnerships and justifications regarding nature conservation (Fisher & Brown 2014).

Thus, it should rather been referred to explicitly as a normative concept (Maris 2014). Particular caution should be exercised while working with this concept marked by ethical controversies. Indeed, while the debate on values remains unsolved, science is unable to contribute in a relevant way: science is threatened by a 'pathological politicization' which would invoke knowledge as a way to settle a conflict on values (Pielke, 2007). At the same time, ES hold the potential to act as *value-articulating institutions* (Martín-López et al. 2014) enabling the inclusion of multiple value domains in a transparent, cumulative and non-substitutive way (Luck et al. 2012). Using them as such in environmental assessments is seen as a relevant way to help society turning toward a sustainable management of social-ecological systems (Kallis et al. 2013).

III. Idealised epistemic commitments: when scientists choose their roles in society

Modern environmental governance relies upon an intense mobilisation of scientists (Coreau et al. 2013). But the extent, the conditions and the objective for which scientists wish to mobilise their knowledge and social recognition remains their personal choice. Characteristics of the various scientific postures embodied regarding governance can be usefully characterised by idealised *epistemic commitments*. Epistemic commitments are defined as "the way scientists combine and "articulate" their research work with issues that matter", i.e. their "commitments both to certain views of knowledge that matters and to certain research practises and networks" (Arpin & Granjou, *in press*). Each commitment is linked to a specific science-society contract but is seldom made explicit and communicated to stakeholders in interaction with scientists. As a consequence, the interface between science and society remains blurry and the objectives of using knowledge for policy are not transparent (Donner 2014).

Some authors have formalised typical epistemic commitments to help scientists in particular but also all citizens to gain understanding of the links between science and governance. The following sections rely on two classifications proposed by Roger A. Pielke (Pielke, 2007: pure scientist, science arbiter, issue advocate, honest broker, stealth issue advocate) and Coreau et al. (2013: guarantor, guardian, officer). The eight scientific postures they defined are hereafter discussed relatively to their overlap with the governance arena, their relative axiological and epistemological contents as well as the visions of democracy and science they support. Figure 2 illustrates the characteristics of these eight typical epistemic commitments.

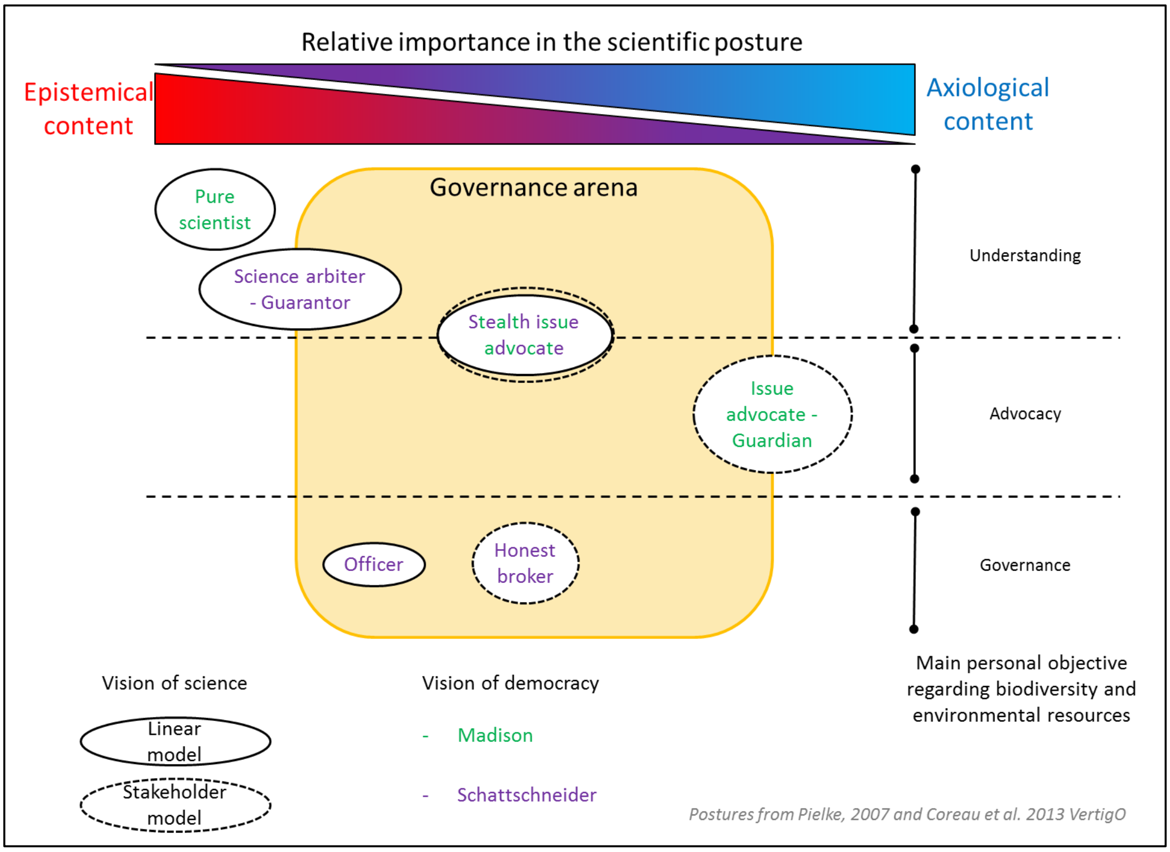


Figure 2: Characteristics of typical epistemic commitments and of their links to governance.

In the specific context of environmental research, the typical epistemic commitments can be defined as follows, according to the description given by their authors (interested readers are referred to their extended definition by their initial author).

A. Pure scientist

The *pure scientist* illustrates the commitment of a researcher focused on knowledge and on gaining more understanding of ecosystems and biodiversity. This researcher places himself/herself outside the governance arena and does not engage in decision-making. Results of research are available within the common pool of knowledge for any stakeholder interested to use them. Questions of values and ethics are not considered as part of the research process, even though respect of deontology is encouraged. In the ES-research domain, pure scientists can be found among those who focus on model and methodological developments. A first example concerns ES modelling issues and scientists that focus on improving modelling capacities, often for a single ES. Among many others, works have been published that focused on the modelling of flood regulation service (Stürck et al. 2014) or on landscape aesthetics (Schirpke et al. 2013), with the explicit objective of increasing practical understanding and technical capacity of modelling. Additionally, methods have also been explored as an objective *per se*. For instance, Lautenbach et al. (2011) analysed the historic changes in ecosystem service supply over the district of Leipzig, East Germany, by using land use data. Their objective was to propose and test a methodology that would “study the development of multiple ecosystem services over more than two time steps and apply techniques different from very simple benefit transfer approaches”. No link with the governance arena is proposed, however the knowledge and methodology gained are made available to who is interested. I hypothesize that a second body of pure scientists in ES research is likely to be found among specific disciplines which focus on particular ‘niches’ of the ES cascade (Haines-Young & Potschin 2010). For instance, researchers from functional ecology would use the concept of

ES but be strongly interested in ecosystem functions and processes (see the connection between research on functional plant traits and ES in Diaz & Cabido 2001 or Lavorel et al. 2011). Acting as pure scientists can obviously be only part of the commitments of these researchers. Overall, I wonder whether being essentially a pure scientist in ES research may be less common than in other domains as by definition ES has been designed to be relevant for calling to mind decision-makers on natural resource concerns.

B. Science arbiter - Guarantor

The *science arbiter* and the *guarantor* hold close commitments. They are experts considering that research does not initially relate to governance as their main interest is to gain understanding. However they recognise that this knowledge can be of help in decision-making and in the management of natural resources. As such, they accept to be consulted by stakeholders to answer *positive* questions (non-*normative* questions, i.e. neutral regarding values), thus placing themselves at the frontier of the governance arena. Science is considered as neutral and intervenes in a technocratic model of decision-making to ensure the scientific validity of the policy options proposed. Researchers are seen as independent and outside value conflicts. I propose that a typical paper demonstrating this posture could be the one proposed by Balvanera et al. (2014) that concludes by illustrating the links between knowledge produced by the authors and governance: "Our analysis suggests that a new generation of research, conducted within the guiding context of IPBES, can inform on the causal chain of links between biodiversity change and ecosystem services. This knowledge is essential if we are to develop a multiscale decision and policy framework designed to effectively manage for biodiversity and ecosystem services over the coming century." Another example of the science arbiter – guarantor commitment is exposed more in details in Section IV: I argue that this is the posture I embodied in the case-study assessed during my PhD (French Alps case-study of the CONNECT project). Indeed knowledge and methodologies explored to increase it were mainly sought for themselves, even though the project asked for results to be relevant for governance at high level.

C. Officer

The *officer* proposes a distinct commitment as its first interest is to favour the use of environmental science within the governance process. This expert is comfortable both with the scientific content of the research and with the institutional functioning of the policy process that can make use of it. This role is supposed neutral and is appreciated for the efficiency it conveys to decision-making, through the sound articulation of knowledge within the political process of negotiation. It supports a linear model of science. The posture of scientists working for the structures in charge of ecosystem assessment at national scale could be characteristic of officers. For instance, the Belgium Ecosystem Services (BEES http://www.beescommunity.be) network is "a community of practice aiming to connect different societal actors involved in ecosystem services research, practice and policy-making" where scientists can contribute to "including methodologies and transfer of knowledge on Belgian ecosystem services to policy". There, scientists can increase the inclusion of knowledge in policy thanks to their position in close connection to decision-makers that, by building understanding and trust, enable them deciphering the institutional system to which they contribute.

D. Honest broker

The *honest broker* has been described as a necessary although challenging commitment that differs from the officer's by its rooting in a stakeholder model of science. Indeed, governance of biodiversity and ecosystems is the core focus of this posture, but the main objective of an honest broker is to expand the range of policy alternatives proposed respective to the various interests of the stakeholders concerned. Science is used to anticipate the outcomes of 'classical' policy options, but also to propose additional options to address the issue under negotiation, even if these options are outside the initial framing of the problem. The honest broker does not advocate for one specific option but helps taking a step back by proposing alternatives. By enabling to 'think out of the box', such commitment could help society avoid the gridlock Einstein warned us against: trying to address problems with the thinking that created them (Barnaud & Antona 2014).

The ES concept was initially proposed to make decision-makers consider nature, as an "eye-opening metaphor intended to awaken society to think more deeply about the importance of nature" (Norgaard 2010). Thereby, this concept was supposed to broaden the scope of governance options and thus to encourage an 'honest brokering of policy alternatives'. In reality, scientists involved in ES research can adopt other postures, as exemplified in this Chapter. Consequently using the concept of ES does not guaranty a neutral and innovative contribution to political debates, even if I believe that it might increase the chances of positive outcomes. One example of scientists willing to act as honest brokers is embodied by the team working on the on-going ESNET project on Grenoble's employment catchment (http://projet-esnet.org/). By building scenarios including ecological and socio-economic data in collaboration with local stakeholders, they will propose alternatives on the future supply of ES modelled at local scale and fine resolution, depending on management choices. A close interaction with diverse and representative stakeholders (including decision-makers), the multiple disciplinary backgrounds of the various researchers involved and the consideration of social and ecological values are key factors in playing an honest broker role in ES research. Whether this team will actually act as such will be assessed through the real outcomes of the project and through their further appropriation by stakeholders, but the team seems off to a good start.

E. Issue advocate and guardian

Both the *issue advocate* and the *guardian* clearly place their interest in advocacy. Research results are used to support specific policy options, generally seeking to promote nature conservation. Thus, the epistemic content of the research is seen as a mean to convince for options in accordance to the scientist's personal values. Expertise is thereby a tool used in a pragmatic manner to support an action of normative basis (Coreau et al. 2013). At the opposite of the honest broker, scientists acting as issue advocate or guardian seek to reduce the range of policy options toward the one they explicitly support. Science is not given as neutral and proximity with conservationists or NGOs does not appear problematic, as engagement and expertise are interweaved and embodied by the same individuals. Many issue advocates have explicitly voiced their position regarding the on-going debate on monetary valuation of ES. For instance, Gomez-Baggethun & Ruiz-Perez (2011) clearly states that "within the institutional setup and broader socio-political processes that have become prominent since the late 1980s, economic valuation is likely to pave the way for the commodification of ecosystem services with potentially counterproductive effects in the long term for biodiversity conservation and equity of access to ecosystem services benefits". An alternative position is proposed by Pavan Sukdev, who led the Economics of Ecosystems and Biodiversity study (TEEB), as he repeated several times in the public press that nature has to be given a price to be conserved, thereby explicitly supporting its economic valuation (Maris 2014). Furthermore, as detailed in Section IV, a close connection to the sphere of environmental activism (e.g. environmental-friendly NGOs) often reveals guardians and issue advocates that will use their knowledge on ES in behalf of their values, for instance in land planning debates.

F. Stealth issue advocate

Finally, the *stealth issue advocate* holds an intermediate and inexplicit posture between understanding and advocacy regarding the use of environmental knowledge for governance. The scientist has a commitment *a priori* disconnected from decision-making and claims his/her main interest in pure science. Results of research are supposed to be neutral and are made available to interested stakeholders. However, the axiological content of the research might not be as minor as presented, and stealth issue advocates usually make use of their legitimacy to advocate for specific options without mentioning it. This posture is typical of situations where science is invoked to solve a conflict of values. This posture is thus embodied by scientists that are not 'naïve' regarding the potential impacts of science in environmental governance but that build an opportunist strategy making use of the linear model of science and its supposed neutrality to drive the decision-making process toward their favourite options (Sandbrook et al. 2011). Coreau et al. (2013) illustrate this posture by hypothesising that the adhesion of researchers in ecology to the mainstream linear political system is rather strategic than naïve. They argue that letting policy-makers the charge of 'formulating the questions' allow researchers to hide behind the science arbiter commitment so as to increase their legitimacy. They describe a fantasized vision of decision-making as a rational choice that would benefit to scientist willing to inexplicitly give weight to their personal values and opinions, thereby acting as stealth issue advocates.

G. Conclusions

With limitations for the stealth issue advocate posture which is characterised by secrecy and inexplicitness, all epistemic commitments described are useful in democracy and contribute to sound relationships between environmental science and society (Pielke, 2007). Individuals can adopt different postures depending on the issue addressed and the step of their career. However, once identified as an issue advocate, a scientist might not be able to present himself/herself alternatively in future debates (Donner 2014). The honest broker posture might be better served by collectives than by individuals due to the broad range of opinions and competences required to broaden the scope of policy options.

What appears important in reviewing such epistemic postures is to understand that science can relate to governance in various ways. In all cases, scientists will adopt a specific posture, be it consciously or not, and that will affect the way society can make use of the knowledge produced.

IV. Reflexive assessment of scientific practises

I previously exposed the notions under debate for the reflexive assessment of research projects linked to environmental resources. This sets the stage for a critical look on the work I contributed to during my PhD and on an additional project that will take over from it. Section IV is conceived as an opportunity to explicit the axiological contents and epistemic commitments of these projects. This should allow me to take a step back on this work so as to communicate more transparently and eventually reframe or adapt parts of it.

A. Projects considered

The first and main project under assessment has been extensively presented throughout this manuscript as it consists in the **French Alps assessment** carried out in the context of the Biodiversa CONNECT project. It first consisted in a quantitative biophysical assessment of interactions between ES and biodiversity, together called ecological parameters (EP) (Chapter I). This assessment was complemented by a qualitative representation of interactions between EP and variables from the social system, by explicitly considering ES three facets and proposing a theoretical framework to map influence relationships (Chapter II). A critical analysis on the governance of issues linked to agriculture, tourism and biodiversity was lastly performed to progress on the understanding of social regulations applied to natural resource management (Chapter III). These three chapters contribute to one of the six case studies that constitute one work package from five in the CONNECT project (cf. Introduction). As such, the following assessment is in no case an assessment of the project as a whole and represents only a partial vision of it applying strictly to the French Alps case study. Therefore, it can be understood as my experience of the case study I contributed to.

The second project under assessment is a follow-up of my thesis that is intended to build upon the results and methodology of my work in CONNECT for a collaborative implementation at local scale. This project has been called **ICARE** (in French for '*Information et Concertation sur l'Aménagement des Ressources Environnementales*') and is conceived as a pilot action research project. It is focused on one community of municipalities in the Mont-Blanc valley. This community, the 2CCAM ('*Communauté de communes de Cluses, Arves et Montagnes*' http://www.mairiedemarnaz.fr/2ccam), showed interest for engaging in this pilot project and therefore will be our main institutional partner. Over its territory, land allocation is highly constrained by the biophysical setting (heterogeneous topography, steep slopes, harsh climatic conditions …). Additional pressures arise from the social system as very high touristic and residential expectations are linked to this iconic area while transport infrastructures and land artificialization increasingly threaten remaining natural and agricultural ecosystems. As such, Mont-Blanc valley expresses numerous tensions regarding land planning and ecosystem management. At the opportunity of a broad audience conference related to the Alps (Université des Alpes, Megève, France – September 2013), we presented the results and potential interests of our biophysical assessment of ecological parameters at regional scale (see Chapter I). Further discussions with a member of a French funding foundation (Fondation de France www.fondationdefrance.org) and representatives from an environmental-friendly NGO (FRAPNA 74 www.frapna-haute-savoie.org) initiated the idea of a joint project that could make use of the methods and scientific insights presented for a local implementation. The ICARE project therefore intends to provide information on ecosystems using the concept of ES in order to raise awareness about the environmental richness of the area. Inclusion of stakeholders is conditional to the project and will shape both the variables assessed and the expected outcomes of the project. By remaining focused on biophysical units and by integrating in a collaborative manner multiple stakeholders, we aim at proposing an alternative framing to land allocation debates that hopefully could help preventing further degradation of natural and agricultural systems. Funding has been partially acquired to date and still need to be complemented. Figure 3 presents the milestones, initial partners and a broad time-frame of the project. A leaflet presenting ICARE has been designed to communicate about the project with potentially interested stakeholders and is proposed at the end of the manuscript in the Appendices from Chapter IV (in French).

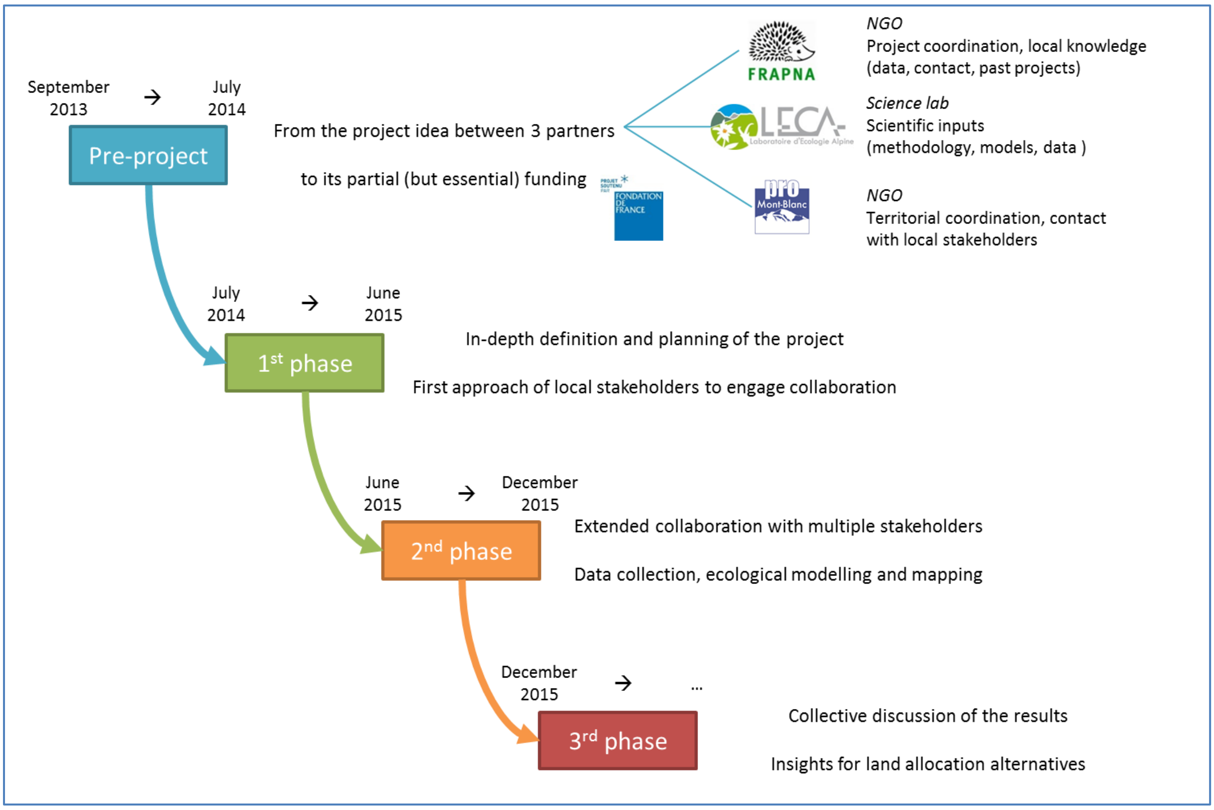


Figure 3: Main milestones of ICARE project to propose a collaborative assessment of environmental resources in Mont-Blanc area.

B. Comparative assessment

Hereafter I discuss important aspects relative to the epistemic and axiological contents of the two aforementioned projects. I propose a sequence of eight questions to communicate on scientific postures and on their consequences on policy-relevant knowledge. I use these questions to assess my work and posture in the context of the two projects.

1. Which vision of science is supported?

CONNECT French Alps case-study	ICARE project
Linear model The case-study has been designed as an occasion to increase knowledge as an objective *per se* and will contribute to a European project intended to inform high-level institutions. I will communicate its results as finalised deliverables to stakeholders involved in the consultative process and additionally to other interested partners. I did not engage a two-way dialog with non-scientific partners, who were mostly regarded as sources of expert opinions.	*Stakeholder model* The project has been conceived to integrate and build from feedbacks between researchers and users of science. The environmental variables I will assess will be collectively decided; data and modelling capacities will come from both academic and non-academic partners. Therefore, knowledge will be co-produced and all stakeholders can be considered as partly responsible for uses of science in decision-making.

2. Which vision of democracy is proposed?

CONNECT French Alps case-study	ICARE project
Schattschneider vision The case-study is intended to deliver recommendations based on the assessment of the alpine system. Our analysis of policy instruments aims for instance at concluding on their efficiency and effectiveness as a mean to inform on their actual or potential effects. As such, we tried to objectively assess the impacts of the actual governance system on biodiversity and ES.	*Madison vision* Debates on land planning options are on-going and our scientific contribution should support environmentally-friendly alternatives. I will mobilize knowledge to support the inclusion of ecological arguments in the political process, which means taking sides.

3. Toward a scientific democracy?

CONNECT French Alps case-study	ICARE project
Not really a- Scientific pluralism has been encouraged as the case-study does not consider pure ecological knowledge. Efforts were made to include social and political sciences even though I finally little collaborated with scientific experts on these domains. b- No thinking on axiological content and value pluralism was developed as a case-study objective or prerequisite. Following a linear model of science, our work was conceived mostly as value-free in the context of the CONNECT project. However, the choices we made in the French Alps case-study (type of assessment, values and indicators) are not value-free even if my ethical setting remained mostly inexplicit to partners external to my team in LECA (see also 7. Axiological background). c- Assessment of chronodiversity is not relevant as we focus on one case-study in a particular time-limited project. As such, I do not think that this project participates to the Slow Science movement in itself. At a personal level, I benefitted from insights of previous and on-going projects carried out in my team.	*Not really* a- The origin of scientific knowledge has not yet been settled but the current framing of the project focuses on one main source of scientific inputs (LECA) and thus on one main approach of science. I will try to make this point evolve once we will have settled on variables to assess. b- Axiological pluralism is obtained by the collaborative approach which characterises this project. Multiple motivations, backgrounds and values are embodied by the various partners concerned. Knowledge and values are strongly related. c- Temporal diversity has not been addressed.

Thereby, the thinking on methods and concepts collectively shaped through years at multiple and complementary spatial scales provided me a very rich background that I could not have accumulated on my own during the time span of my thesis. I believe that this collective heritage pertains to a kind a slow science, or at least to a science balancing short-term with mid- and long-term concerns.	

4. How is public participation considered?

CONNECT French Alps case-study	ICARE project
Consultative process In the case-study, I collected a huge amount of information from multiple stakeholders of regional expertise. Their inputs represent diverse viewpoints without being exhaustive. The process was time-consuming but delivered me essential information for assessing interactions among the social-ecological system and for relating scientific findings to current governance choices. I delivered feedbacks to participants to inform them about the results but no interactive dialog was launched to discuss the research process and its objectives. No conflict or tensions among stakeholders arose as our results would not impact them on-the-ground.	*Collaborative process* The process intended is both time-consuming and expensive but its legitimacy relies upon a broad adhesion and collaboration from multiple stakeholders. Indeed, the project can have political relevance as both the collaborative process and its outputs could be used to support specific land planning options. The inclusion of varied stakeholders is compulsory, which might lead to conflictual demands and expectations within partners of the project. However, degree of involvement will vary, with the three initial partners and the community of communes as main holders of the project and of its outcomes. I must be really cautious to propose credible outcomes of the modelling and mapping process so as not to deceive partners, by making clear the assumptions, limits and uncertainties linked to the models and concepts used. However, this project is neither prescriptive nor exhaustive on land planning aspects, thus we will present it as a collective contribution to territorial management focusing on natural resource issues (among others that need to be considered in land planning). As such, I make very clear that this project does not intend mixing with, nor replacing, the official decision-making process. This precaution, supported by all partners, should help preventing oppositions to the project.

5. How to deal with uncertainties and with a science in-the-making?

CONNECT French Alps case-study	ICARE project
No particular attention except definitions - I did not deal with uncertainties in particular in the case-study. Quantitative results were not associated to levels of confidence as no data or methodology was available to assess it. However, we discussed causes of uncertainty as well as its potential consequences on the reliability of our results and their interpretation. Uncertainty assessments are not yet part of the common culture despite their interest. - We paid particular attention to clearly define the concepts mobilised in the French Alps assessment as they are not yet stabilised (e.g. trade-off, ES facets).	*Point of attention!* - A question unsolved to date is how to communicate uncertainty and validity of the results to the broad range of stakeholders potentially interested. I believe that communicating on uncertainty remains a point of high attention if we want the outcomes of the project to be used relevantly and not distorted toward inappropriate interpretations. - Definitions and concepts will be defined explicitly so as to reach consensual terminology and phrasing among project partners.

6. How is the ES concept considered?

CONNECT French Alps case-study	ICARE project
Concept to deal with complexity ES were considered by scientific partners as a concept useful for exploring the complexity of social ecological systems. In the French Alps case study, we used the concept to include multiple languages and value-domains (quantitative and qualitative assessments – ecological and social aspects included). Overall, I used the ES concept with an academic vision, even if the study has intention of policy relevance. At the beginning of the consultative process, we feared that some stakeholders could get opposed to our requests due to controversies attached to the ES notion. Thus we adapted the terminology we employed in the first steps of consultation and talked about 'environmental resources' and 'natural resources and functions of ecosystems'. We did not find it necessary for the last stages of the study and explicitly mentioned ES. I mostly relate this change to an increased public acceptability and understanding of the concept among the experts we worked with.	*Boundary object* *+ Conflictual theoretical concept* Project partners initially saw ES as boundary objects that can be used by multiple stakeholders. We believe ES have the potential to displace the debate on environmental resources on alternative and hopefully less conflictual domains (in comparison to the "conserve or urbanise" opposition). But, as faced in the French Alps case-study, ES could also give rise to tensions due to their widespread economic and utilitarian framing that can prevent stakeholders from engaging in the project. As such, it is not sure that our assessment will mention ES directly. We might rather focus on terminology as 'environmental resources' and 'ecological profiles' so as to remain more integrative.

7. What about the axiological background?

CONNECT French Alps case-study	ICARE project
Not made explicit Science in the case study is seen as neutral. We produced knowledge *per se* and not for normative aspects. Values and ethical stances were not explicitly discussed. However, the languages of valuation we chose in the case study were integrative (biophysical values – inclusion of social aspects in qualitative terms) and did not focus on unique aggregated indicators. Thus complexity was acknowledged.	*Point of vigilance!* The common values shared by the three project initiators are environmentally friendly and favour conservation of natural and agricultural areas in land allocation. Yet, the project is open to engaging with stakeholders with other concerns and value backgrounds. The Human-Nature metaphor is not made explicit and values of biodiversity are not debated (intrinsic – instrumental…) as the focus is more on pragmatic and readily useful outcomes. I stress that care must be exercised regarding the use of scientific arguments in a political process. As such, this politicization of science should be transparent and explicit so as to avoid stealth issue advocacy in a debate where confronting multiple values.

8. What is the main epistemic commitment proposed?

CONNECT French Alps case-study	ICARE project
Science arbiter - Guarantor The overarching objective of the case-study is to increase understanding. Thus, the epistemic content of the process remains our focus, even if there was no reluctance to engage with governance aspects and to answer positive questions. The previously described combination of a linear vision of science and a democracy conceived after Schattschneider are characteristic of this 'Science arbiter – Guarantor' commitment.	*Issue advocate - Guardian* Both the axiological and epistemic contents are important for this project as values initiated this collaboration while science legitimated it and was seen as a support for decision. The link with the governance arena is intimate even if not correlated to any official process. Our main concern as project partners is to propose alternatives to the consumption of natural and agricultural areas, which is typical of guardians or issue advocates. I stress that this commitment participates to the will of widening the scope of policy alternatives currently considered, although if alternatives will remain coherent with our main advocated position. Finally, I warn against the stealth issue advocate commitment that could be negative for all partners and backfire to the scientific partners.

This comparative assessment on the work I contributed to can be considered as a reflexive exploration of values and presuppositions (ethics *and* science, 'generic ethics' (Coutellec, 2012b)). Even if the two projects are presented separately, they share mutual influences. My contribution to ICARE is conditional to and enriched by my experience from CONNECT at a scientific level (methodological and conceptual background) as well as at a personal level (exploration the controversies and interests of the ES concept – understanding of stakeholder participation and of governance). Conversely, the first steps of ICARE made me more critical (regarding both positive and negative aspects) on the roles of science in governance, and helped me progressing in the reflexive assessment of my work, which I needed to feel comfortable with CONNECT case-study assessment. At the level of the scientific team in which I work, I hope that the outcomes of ICARE will contribute to our thinking on uses of ES research and will exemplify an *action research* orientated process.

No judgement is made regarding the epistemic commitments I endorse (i.e. no one is 'better' than the other in absolute terms) as pluralism in methods, opinions and postures is required to progress toward integrative and citizen sciences. I conclude from this experience that scientists should be encouraged to engage explicitly in this kind of reflexive assessment so as to be conscious of the relationships they favour with society.

The sequence of eight questions I answered for the two projects could be used as guidelines for all researchers interested in communicating in a concise and yet explicit way how they conceive their contribution to governance. Additionally, I refer interested readers to the recent paper of S. Donner (2014), who proposed a similar list of 9 questions he advises "to review when choosing a position along the [science-advocacy] continuum". He proposes three themes to progress in making commitments explicit: i) "choose a place that is right for you", ii) "consider whom you represent", and iii) "analyse your strengths and motivations". Both frameworks could be usefully transposed to apply to non-academic stakeholders, whose commitments and value backgrounds should also be exposed in order to build transparent partnerships.

V. Synthesis

As a general conclusion, Figure 4 summarizes my approach of ethical concerns and proposition to address them.

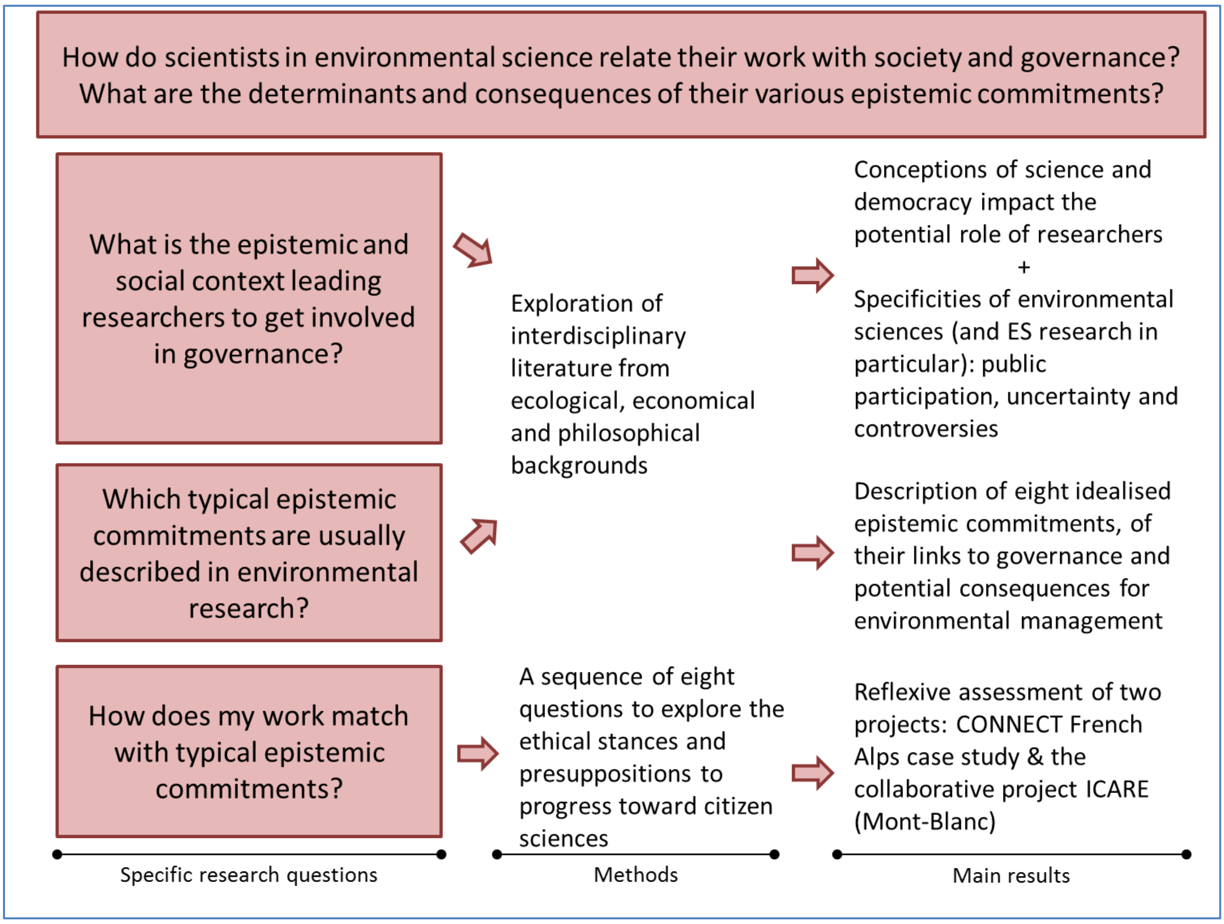


Figure 4: Specific research questions explored regarding the conceptual and ethical issues linked to research in ES domain (Chapter IV), related methods and main results obtained.

This chapter aimed at presenting the questions I faced during my PhD project and the insights I came across by exploring the literature in the ecology domain and in other disciplines. As a personal progression, Chapter IV does not address all ‘hot topics’ linked to the ES concept and its applications and mostly focuses on the links between science and society. I consider the exploration of epistemic commitments and the sequence of questions proposed above as necessary steps to personally undertake so as to better anticipate the tensions that environmental scientists can face. This exercise can be repeated and adapted to very different projects in order to take better advantage of collaborations and opportunities in the academic and non-academic spheres. I also hope that this thinking, among others, could be of help for students (and others) entering the ‘ES arena’. It could support them in their will to better understand the concept they are to work with as well as the options they have for communicating with stakeholders and for making their science relevant.

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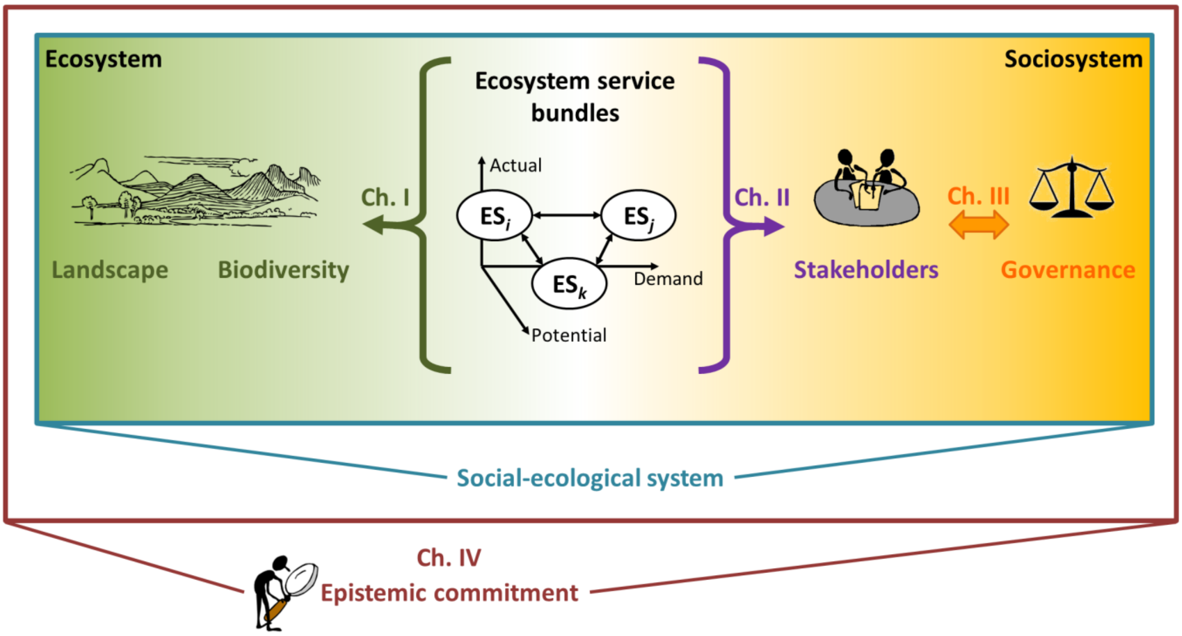
Discussion


Ecosystem
Sociosystem
Ecosystem service bundles
Actual
Demand
Potential
Landscape
Biodiversity
Ch. I
Ch. II
Ch. III
Stakeholders
Governance
Social-ecological system
Ch. IV
Epistemic commitment

Discussion

I. What has been achieved? – Synthesis of the chapters

The overarching objective I pursued during my PhD was to explore the determinants of relationships among ecosystem services (ES) and biodiversity (together referred to as ecological parameters) and the subsequent impacts for their joint management. To benefit from the specific interface position of the ES concept (Haines-Young & Potschin 2010, Martin-Lopez et al. 2014), my approach combined insights from ecological sciences and social sciences and relied both on quantitative and qualitative methods.

Through the four chapters that constitute this manuscript, I proposed different perspectives on the trade-offs and synergies among ecological parameters in the particular setting of the French Alps social-ecological system. To account for the multiple interrelations among ecological parameters, I specifically relied on the concept of ‘bundle’ to describe consistent associations in space and/or time (Raudsepp-Hearne et al. 2010).

In Chapter I, we modelled and mapped 18 ecological parameters assessed through a biophysical approach. We determined their spatial bundles using a pattern-based approach of multifunctionality and highlighted their links to landscape features. We concluded on the overall high ES supply over the French Alps, characterised three regional bundles of ecological parameters and described five sub-regional clusters supplying consistent sets of ES. Finally, we proposed that multifunctional landscapes could be both heterogeneous (e.g. rural mosaic) and homogeneous (e.g. alpine forests) and are conditional to non-intensive practises explicitly targeting multiple objectives.

In Chapter II, we proposed a new conceptual framework, the Influence Network framework, to qualitatively describe the influence relationships among components of the social-ecological system. This framework specifically accounts for ES three facets (potential supply, demand and actual supply) and aims to unravel the network of influence variables and impacted variables around target ES. We based our exploration of the French Alps system on a four-step consulting process with stakeholders of regional expertise. Our results suggested that stakeholders perceived a prominent influence of social variables (e.g. land allocation choices – demands from specific stakeholders groups as leisure hunters) and highlighted that management generally targets provisioning and cultural ES, at the expense of regulating ES and biodiversity.

In Chapter III, together with a Master student whom I co-supervised, we tested a methodology proposed by our CONNECT partners to assess the environmental effectiveness of policy mixes. We focused on a specific bundle of ES to explore the governance instruments currently used to manage the influence relationships at the interface between agriculture, nature tourism and biodiversity. Our extensive literature review was complemented by a set of individual interviews to assess the individual environmental effectiveness of ten instruments within the policy mix. We further characterised the broader governance network of these instruments to progress in the understanding of their positive complementarities and negative overlaps. We concluded rather positively regarding the environmental effectiveness of the policy mix and clustered instruments to characterise usual mechanisms of articulations through scales and sectoral domains of interest. We highlighted the interest of exploring ‘rebound effects’, i.e. untargeted positive and negative collateral effects of policy instruments. In particular, we warned against potential negative spatial spill overs (e.g. differentiated agricultural management modalities according areas for the geographical indications) and promoted potential synergies arising from widespread positive impacts on all ES categories (e.g. positive impact on cultural services from the support of extensive agricultural practises through agro-environmental schemes). A policy brief communicates our general conclusions, with stakeholders at regional level as our primary target audience.

In Chapter IV, I presented the conceptual and ethical questions about ES sciences I came across during my PhD. I described the science-policy interface and proposed eight typical epistemic commitments describing how environmental scientists can make their science relevant. I additionally proposed a sequence of eight questions to make epistemic and axiological stances more explicit. I concluded on my epistemic commitments over the (short!) timespan of my PhD and encourage all stakeholders (i.e. not only scientists) to personally undertake this kind of exploration so as to favour transparency and explicitness.

Figure 1 proposes a synthesis of these chapters and of their main results.

Figure 1: Specific research questions explored to approach the trade-offs and synergies among ES and biodiversity in the particular setting of the French Alps social-ecological system, related methods and main results obtained.

II. What can we conclude about the French Alps social-ecological system? – A subjective description

Our approach sequentially analysed bundles of ecological parameters accounting successively for their biophysical patterns (Chapter I), their ecological and social influence networks (Chapter II) and the policy mix that manage their supply (Chapter III). By integrating the insights of this multi-layered description, I now present a synthetic overview of the French Alps social-ecological system. It summarises and interweaves some striking features that were further detailed in the previous chapters. My objective here is thus to re-integrate the different perspectives I individually exposed so as to present one (personal) vision of the alpine social-ecological system.

A. A description of the alpine system through some 'visible' ES

I previously presented the consultative process we carried out to describe the alpine social-ecological system (Chapter II). During this process, the management of natural resources was characterised by considering the ecological and social influence relationships around numerous ES. Yet, some specific ES were more actively discussed by our stakeholders than others. Further, some ES were explicitly prioritized in the third step of the process when we built detailed influence networks around ES chosen by stakeholders according to their (justified) 'relevance' in the alpine setting. Following the terminology by Lamarque et al. (2011), I hypothesise these ES have a 'visible' structuring effect on landscapes and natural resource management. I hereafter propose to build from this restricted bundle of ES to describe some features of the alpine system that I believe important. Although not selected through a robust ranking process, the set of ES discussed is proposed as an entry point describing three issues I find critical for sustaining ES supply, in relation to i) an increasing demand for some ES (natural hazard mitigation, nature tourism and wood and wood energy production), ii) the critical importance of the French Alps area as a water tower (fresh water regulation and hydro-energy), and iii) an increasing pressure on some ES (erosion mitigation, agricultural productions and landscape aesthetics) (Figure 2). The following sections highlight some biophysical relations underlying bundles of ES and further include the influence of the land use matrix and the history of management intensity on the presented bundles.

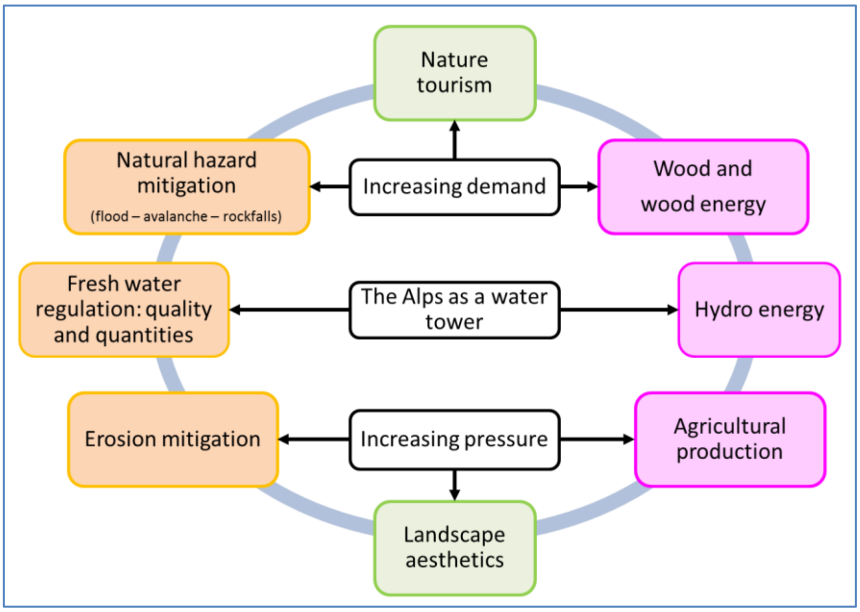


Figure 2: Subjective selection of ES presenting a structuring visible effect on the alpine social-ecological system. ES categories are symbolised by colour backgrounds (pink = provisioning; green = cultural; orange = regulating).

1. An increasing demand for specific ES

First, I highlight three ES that are currently experiencing increasing social demand: nature tourism, wood and wood energy production, and natural hazard mitigation.

Nature tourism represents a prominent economic asset for many alpine areas (POIA 2014) and builds both on biophysical mountain specificities (e.g. slopes, climate…) and on cultural landscape aesthetics, the latter being therefore critical to maintain. Outdoor practises have both increased and diversified over the past 25 years and this trend is foreseen to continue. The stakeholders we consulted described the various environmental impacts induced in particular by mass tourism (e.g. urban and infrastructure development, water cycle alteration, wildlife disturbance) and mentioned the importance of regulatory instruments to limit them (e.g. UTN – authorisation procedures – impact studies). Sustaining the nature tourism service leads to trade-offs with other ES, in particular from the regulating category. Our spatial congruence analysis demonstrated the suitability of mosaic grassland landscapes for outdoor tourism and therefore supported the consideration of rural tourism opportunities in the extensive agricultural sector. This was confirmed by our governance analysis which proposed instruments targeting forms of 'soft' tourism to sustain both economic aspects in general and the agricultural sector through rural forms of tourism in particular or specific local products, from high policy levels (e.g. Tourism protocol of the Alpine Convention) down to very local pilot projects (e.g. 'Alpe en Alpe' agro-tourism project). Overall, managing the tourism opportunities developed to answer the increased demand for nature tourism is a critical future challenge. I believe efforts should focus on mitigating mass tourism impacts while promoting alternative (and economically more redistributive?) forms of soft tourism such as agro tourism or small-scale rural tourism.

Alpine forest products for timber or fuel are overall considered as an under-exploited potential to date (CMA 2006). The current conjunction of an increased demand for renewable energy and for local materials presupposes a higher actual supply of wood production in future years. They are supported by current policy orientations from the '*Law on the energy transition for a green growth*' that favour the use of renewable energy sources. The stakeholders we consulted encouraged the use of geographical and quality indications for alpine forested products so as to promote both profitability and sustainable practises (e.g. PEFC, '*Bois des Alpes'*). We showed the multifunctionality of alpine forests through our spatial congruence analysis which identified a consistent bundle of forest-related ES. Thus, satisfying the increasing demand for forest products should be done by maintaining a careful attention to the multifunctional objectives already pursued by alpine forestry, in particular natural hazard mitigation, fresh water regulation (quality and quantity) and carbon storage. The stakeholders consulted stressed the need for particular attention to fire risk mitigation, notably in the Southern Alps and in the context of climate change. Despite favourable biophysical conditions overall, the economic profitability of forestry remains conditional to an adequate network of forest servicing (in terms of both access and transformation), which was highlighted as an important issue during our consultative process. Moreover, we stressed that relations between forest managers and other stakeholder groups (hunters – tourists) are a sensitive social and political issue, as they reveal the numerous and potentially conflicting demands toward forested ecosystems. In particular, there is a need to address the tensions between forest managers, hunters and naturalists regarding wild ungulate abundance and their impacts on forest regeneration. I see in the increased demand for forest products an opportunity to bring together the various stakeholders concerned by forested ecosystems so as to collectively manage these areas of high multifunctional potential. Forest-related stakeholders from multiple organisations have already begun exchanging views and knowledge in informal networks such as the Ecological Forest Network of Rhône-Alpes region (Réseau Ecologique Forestier Rhône-Alpes - REFORA http://refora.online.fr/) that bring together for instance forest owners, forest managers, scientists, nature conservation organisations and natural areas users. This example appears instructive to collectively tackle the challenge of sustainable forest management.

The French Alps are highly exposed to natural hazards due to their "geomorphology with high mountains, deep valleys, permafrost areas and glaciers in combination with events of heavy rain- or snowfall and exceptional gradients of day and night temperature" (ClimChAlp 2008). Increasing urbanisation of valleys and tourist numbers reinforce the demand for mitigation of natural hazards, and in particular for the protection against floods, avalanches and rockfalls. Ecosystems can supply a biotic contribution to limiting these risks. For instance, our biophysical assessment considered rockfall hazard and proposed a specific cluster supplying mitigation of this risk. The other ES supplied by the characteristic bundle of this cluster related to the same ecosystem, i.e. forests (carbon storage, wood production), and to its abiotic particular conditions, i.e. steep slopes (hydro-energy production). Natural risk prevention plans were mentioned during our consultative process as priority instruments at municipal level for implementing preventive action, in accordance with higher-level policy objectives (e.g. POIA 2014). Prevention plans are concerned by land allocation choices and resulting land cover types. More specifically, their objectives are to sustain protective ecosystems and to limit population exposure (e.g. by adequate planning of urbanisation areas or by maintaining favourable practises such as grazing of high altitude grasslands for avalanche prevention). In addition, the stakeholders we consulted insisted on the spatial upstream-downstream dependence regarding flood mitigation and on the resulting need to maintain attention to this risk even at distance from its spatial source. Consequently, I believe that environmental management in the French Alps cannot bypass natural hazard mitigation, in particular in the context of climate change that will exacerbate risks (Grêt-Regamey et al. 2008, Elkin et al. 2013). I also stress the importance of considering spatial dependencies and links between tripping and exposure zones beyond administrative municipal borders.

2. The French Alps as a water tower

The second issue that I propose for describing the alpine social-ecological system deals with water-related ES (hydro-energy production – maintain of water quality and regulation of water quantities). Water in the French Alps is considered overall as an abundant resource and yet remains fragile and unevenly distributed (CMA 2006). The region has been called 'a natural water tower' regarding the numerous rivers having their headwaters there and irrigating lower areas well beyond the massif borders (Körner & Spehn 2001, Viviroli et al. 2007, EEA 2009). However, multiple water uses compete (agriculture – industry – energy production – tourism – drinking water …) and tensions are rising in the context of climate change (Schädler & Weingartner 2010).

The French Alps hold a great potential for hydro-energy production and have been highly equipped in hydropower plants, up to supplying half of the national hydro-energy production (CMA 2006). However our consultative process revealed conflicting demands affecting its actual supply. On the one hand, the increased call for 'clean' sources of energy favours the actual supply of this ES. In this context, the development of micro electric plants is promoted in governance (e.g. CMA 2006 - Law on the energy transition for a green growth). Local synergies with nature tourism (water sports – artificial lakes as hiking destinations…) were highlighted. On the other hand, hydro-energy production was mentioned to decrease landscape quality, to alter water cycles and to disturb environmental quality, in particular due to the sedimentary misbalances caused by dams. Policy instruments have been designed to limit these negative impacts (e.g. the national 2006 law on water and aquatic environments). Overall, I believe that major infrastructures are to be considered as quasi-fixed constraints supplying renewable and local energy. Management should favour synergies with tourism and target the maintenance of ecological and sedimentary continuities. Alpine ecosystems' contributions to this service could also be further fostered by an increased consideration of vegetation cover at the watershed level, as discussed in our biophysical analysis (Chapter I).

Management of fresh water is needed to sustain its overall good quality (CMA 2006). Our spatial congruence analysis demonstrated the necessity of promoting ecosystem retention capacities (of nitrogen but also of other pollutants) in particular in agricultural areas. Indeed, these areas are exposed to a quite high pollution risk while offering potentially large vegetation cover areas able to fix the nutriments and pollutants before they can reach water bodies (excepted for bare soils of annual cropping). Our consultative process suggested the potential contaminations from livestock farming in particular at basin heads, even under extensive management conditions. Moreover, stakeholders described the negative impacts on natural purification capacity from the increased population in the valleys, due to both additional pollution sources and to a decrease in perennial vegetation covers in urbanised areas. Our governance analysis incorporated instruments designed to include environmental concerns in agricultural practises (e.g. CAP II Pillar) and also to control urbanisation impacts on water quality (e.g. Water Framework Directive). The maintenance of water quality appeared linked to regulation of water quantities, although the two ES were not spatially congruent in our biophysical analysis. Indeed, high regulation of water fluxes was linked to forests, which play an important buffering role, while quality was rather overlapping with agricultural areas, due to their higher exposure. Soil sealing by urban and infrastructure development was mentioned by the stakeholders we consulted to impede water infiltration and thus to induce higher erosion rates and flood risks. Water-related services were not a particular focus of our study (partly because of a lack of available expertise) but I nevertheless stress the importance of their full consideration to manage and allocate sustainably environmental resources.

3. Highlights on some ES undergoing increasing pressures

Third, I propose three 'visible' ES that appeared to undergo an increasing pressure and whose management remains therefore a future challenge: erosion mitigation, agricultural production and landscape aesthetic value.

Erosion mitigation was frequently discussed during our consultative process. This "matter of primary importance in mountain areas" undergoes an increasing pressure resulting from "increasing numbers of tourists, changes in farming/cultivation techniques and climate change" (Bosco *et al*. 2009). Stakeholders proposed both landscape composition and configuration as important drivers of erosion mitigation. Our spatial congruence analysis indeed related this ES to forests functionally able to retain soil losses and also to mosaic landscapes where, consistent with landscape ecology literature, I hypothesise land configuration to play a positive role (see for instance Syrbe & Walz 2012). The success of past voluntary actions to mitigate erosion was acknowledged for natural and agricultural areas (plantation of protective forests, terraced land arrangements, action from the 'land restoration service for mountain regions'). However, urban sprawl is increasing soil sealing and limiting infiltration. Stakeholders mentioned the increased appearance of small mudflows on mountain villages as a result of this trend. They also described positive measures to mitigate the direct impacts of tourism in sensitive areas (e.g. works of the National Forest Office to restore busy trails prone to erosion). Additionally, stakeholders appeared concerned by the intensification of agricultural practises (e.g. increase of carrying capacity in pastures - deep ploughing – bare ground in winter) and by the changes in pastoral practises due to the presence of wolf (concentration of herds in night enclosures). They stressed the subsequent potentially large losses of fertile soils and their further negative impacts on water quality, consistent with findings in the literature (e.g. Bakker et al. 2008 for impacts of agricultural changes on erosion rates in rural mountain landscapes). At least four causes of erosion are likely to intensify in future years: climate change, changes in agriculture practices, tourism and soil sealing. Thus, I stress the relevance of considering erosion mitigation as a critical challenge whose management will require considering both its social and ecological drivers.

Agricultural production is a structuring activity in the French Alps regarding the landscape, the economy and the culture, although unevenly distributed and characterised by varying management intensities. Our consultation process identified this ES as the basis of the strong identity of the territory and of its cultural and attractive landscapes. The congruence analysis revealed the joint potential of alpine grasslands and pastures for supplying recreation and tourism opportunities as well as fodder production. We also stressed the biophysical potential of agricultural areas to provide habitats and resources for many plant and animal species. Several ES from all categories were described by stakeholders as i) being conditional to the actual agricultural production supply, and ii) being directly impacted by the management of agricultural areas. As an example, stakeholders mentioned provisioning services naturally, but also cultural services such as landscape aesthetic or educative value and regulating services such as pollination, maintenance of water quality or erosion mitigation. Our governance analysis explored many policy instruments from all categories dedicated to managing jointly these ES: regulatory (e.g. national '*Ecophyto*' plan to limit contamination), economic (e.g. agro-environmental measures from CAP II Pillar) and voluntary instruments (e.g. geographical indications). However, strong pressures are threatening the livelihood of mountain agriculture, including changes in markets and governance, abandonment of agricultural lands, presence of large predators. Our consultation stressed that management of this complex bundle of ES is strongly challenged by an increased pressure from land allocation choices and in particular from urban sprawl. Governance is addressing this issue through numerous instruments trying to protect agricultural areas (e.g. protective perimeters) and to balance land allocation (e.g. regional ecological coherence scheme - local urban development plan). But from our consultative process, we conclude that land allocation conflicts remain prominent and highly challenging to address. Overall, I believe that successfully maintaining extensive practises favourable both to biodiversity and several ES remains conditional to supporting farmers both spatially (land planning choices), economically (decent income), socially (addressing new social demands in terms of facilities and time management) as well as in terms of supporting expertise and knowledge.

Although not addressed by our biophysical assessment, landscape aesthetic value was frequently discussed by our stakeholders. It is also increasingly considered by governance, as exemplified by recent "atlases of landscapes" at the 'département' level supporting a multifunctional management of natural resources. In our consultative process, landscape quality was described as a major visible output of the actual encounter of biophysical supply and conflicting social demands, regarding both past and current uses. Almost all ES were related to landscape quality, which was further proposed as a very relevant entry point for increasing public awareness and understanding of environmental management. The aggregated consequences of changes in ES supply and in social demands were considered during our consultative process. They conveyed a negative projection for landscape quality, in particular due to woody encroachment and landscape fragmentation.

B. Insights from contrasted opinions on biodiversity

Throughout our analyses, we explored the interactions between ES and biodiversity. I propose four stances to approach the social-ecological system from the ways the stakeholders we consulted mentioned their relations to biodiversity. The following results browse the diversity of concerns they expressed. These are not exhaustive but I nevertheless believe they are interesting entry points to approach the alpine social-ecological system.

1. Biodiversity as "the impacted variable"

During our consultative process, most stakeholders referred to biodiversity as the "impacted variable". They appeared well aware of the consequences of human activities and of uses of ES on biodiversity. Impacts arise from three factors: tourism, urban sprawl and the actual supply of provisioning ES. There is a clear trade-off between the higher direct profitability linked to intensive practises and their negative impacts on biodiversity. Stakeholders are then faced with "the requirement to limit their impacts on biodiversity" that can either be inspired from personal feelings or imposed by formal institutions. Four points arose from our analyses in relation with this conception of biodiversity.

First, in our congruence analysis, we pointed out the potential suitability of agricultural areas for plant diversity and clearly mentioned that the actual presence of diverse plant species remains conditional to an agricultural management of low or intermediate intensity. We explored some related policy instruments (e.g. agri-environmental measures – voluntary programs such as '*Agrifaune*') and concluded on their overall synergistic articulation in the policy mix.

Second, consistent with our biophysical analysis, our stakeholders judged negatively the relationship between hydro-energy and animal diversity due to the ecological discontinuity this ES induces. Both legal and voluntary instruments are proposed in the policy mix we explored to address this issue (e.g. law on water and aquatic environments - charter of good practices for energy infrastructures).

Third, we showed the suitability of forested ecosystems for hosting vertebrate species. This relation echoed with the conflicting management of wild ungulates species damaging through their intense browsing these habitats which supply wood products.

Finally, we found that potential habitats favourable to plant diversity partly overlapped with areas currently dominated by artificial covers. Thereby, we highlighted the need for compromises in land planning choices, reinforced by the frequent mention of biodiversity as a 'strong constraint' for urban planning and infrastructures. Our analysis considered several regulatory policies designed to assess and control the impacts of large development projects on biodiversity, ecosystems and ES (e.g. regional ecological coherence scheme – UTN procedure – mountain law – protective perimeters – measures to 'avoid, minimize or compensate' impacts).

Additionally, landscape composition and configuration was mentioned by stakeholders as a major driving factor of biodiversity. This was also acknowledged by our governance analysis which explored the implementation of green and blue corridors through multiple instruments operating at nested scales (e.g. Grenelle's laws at national scale, the scheme of ecological coherence at regional scale, the 'Biological corridor' program at the scale of the 'département' and local perimeters of protections as Natura2000 or ENS).

Overall, biodiversity impacts were often presented as collateral effects of practises targeting other objectives (tourism opportunities – urbanisation – provisioning ES). To limit them, three mechanisms were mentioned. First, some stakeholders spoke about informal institutions (i.e. norms, values) linked to a cultural alpine identity which would be *in essence* mindful of these aspects. However, not all stakeholders were convinced that these informal institutions actually had positive effects on biodiversity conservation. Second, the policy mix appeared well instrumented to control the impacts of human activities with instruments from all categories (regulatory, economic and voluntary). Third, stakeholders mentioned the interests of entering the social-ecological system through a landscape perspective. Indeed, landscape was presented as the result of combined impacts from various drivers (e.g. agriculture, social demand, urbanism…) which are often possible to manage.

2. Biodiversity as a factor of attractiveness

During our consultative process, a distinct opinion was frequently mentioned by the stakeholders we consulted. Biodiversity was also referred to as an attractive feature highly representative of alpine territories and of their overall good environmental quality. Much of the alpine cultural identity conveyed for attracting tourism relates to the high levels of biodiversity and to correlated environmental quality (e.g. national park communication – specific public events organised by the National Forestry Office).

Some stakeholders noted that biodiversity *per se* was not such a strong factor of attractiveness. This was also highlighted in our spatial congruence analysis where plant diversity and nature tourism were linked to distinct bundles at regional scale. An output of the consultative process is that the general public seems rather to focus on specific endangered or visible species (e.g. large predators – 'nice' flowering plants). Iconic species play indeed a particular role as they are often put forward to justify the perimeters protected (e.g. Natura2000) and the sites promoted (e.g. plans for tourism trails and sites designed at the 'département' level), at the expense of more common or less visible species. For instance, the stakeholders we consulted mentioned that the return of the wolf in the French Alps had been promoted as a marker of wilderness in the tourism sector. Yet, some stakeholders specified that 'wilderness' is positively perceived when it remains 'human-managed', as for the presence of wolves in the Alpha wolf centre, Southern Alps, echoing to the "*Canada Dry* wilderness" evocated by Larrère (1994).

Some stakeholders also mentioned the dangers of focusing on a restricted list of species to design conservation and assess impacts: they highlighted the low representativeness of such species regarding the broad range of resources and habitats required to sustain biodiversity in general. This was particularly salient when we constructed our indicator of iconic species in the spatial congruence analysis. It stresses the political significance of the allocation of greater attention to certain species and habitats.

Overall, biodiversity as a whole or considered through particular species appears strongly and positively linked to alpine cultural identity. It is therefore positively related to numerous cultural ES (tourism and recreation – educational value…). The challenge is now both to protect the natural habitats hosting these species and to sustain the practises that shaped the cultural landscapes to which they are adapted. The latter are in particular the extensive agricultural practises that are widespread in the French Alps but that are also threatened by global changes (Lamarque et al. 2011, Tscharntke et al. 2012, Noury & Poncet 2013).

3. Biodiversity as an insurance

Some stakeholders we consulted demonstrated a rising awareness regarding the higher potential of diverse ecosystems to face changing conditions. In particular, the stakeholders with forest expertise proposed that heterogeneous forests in terms of ages and species would be less sensitive to extreme summer droughts, violent storms or new diseases. Then, short-term profitability (that can be higher in forests dominated by one or two species) was balanced with mid- to long-term sustainability of the ecosystem. Forest experts and managers increasingly consider the management of forests as a way to promote its adaptive capacity. However, stakeholders of other sectors of activity did not mentioned biodiversity under this perspective, with an exception for the management of extensive agricultural areas faced with climate change.

4. Biodiversity as an essential functional support

I already highlighted that biodiversity was mostly considered in terms of impacts, meaning that thinking about biodiversity in terms of its influence on the supply of ES was not straightforward for the stakeholders we consulted. In other words, the actual contribution of biodiversity to ecological processes and further to ES was not spontaneously highlighted during our consultative process.

However, scientific evidence of the impacts of biological diversity loss on the functioning of ecosystems and their ability to supply ES has been recently gathered. A review by Cardinale *et al.* published in *Nature* (2012) proposed to synthetize robust findings in 6 consensual statements (directly quoted from the review):

1. “There is now unequivocal evidence that biodiversity loss reduces the efficiency by which ecological communities capture biologically essential resources, produce biomass, decompose and recycle biologically essential nutrients.”
2. “There is mounting evidence that biodiversity increases the stability of ecosystem functions through time. “
3. “The impact of biodiversity on any single ecosystem process is nonlinear and saturating, such that change accelerates as biodiversity loss increases.”
4. “Diverse communities are more productive because they contain key species that have a large influence on productivity, and differences in functional traits among organisms increase total resource capture.”
5. “Loss of diversity across trophic levels has the potential to influence ecosystem functions even more strongly than diversity loss within trophic levels.”
6. “Functional traits of organisms have large impacts on the magnitude of ecosystem functions, which give rise to a wide range of plausible impacts of extinction on ecosystem function.”

Regarding the supporting role of biodiversity for ecological functions and further for ES, there is thus a discrepancy between the scientific evidence and the perception of the stakeholders we consulted, which is consistent with other studies (e.g. in a Mediterranean semiarid region - Castro et al. 2011). Integrating robust findings about biodiversity in the common environmental management culture remains a current challenge.

5. Synthesis

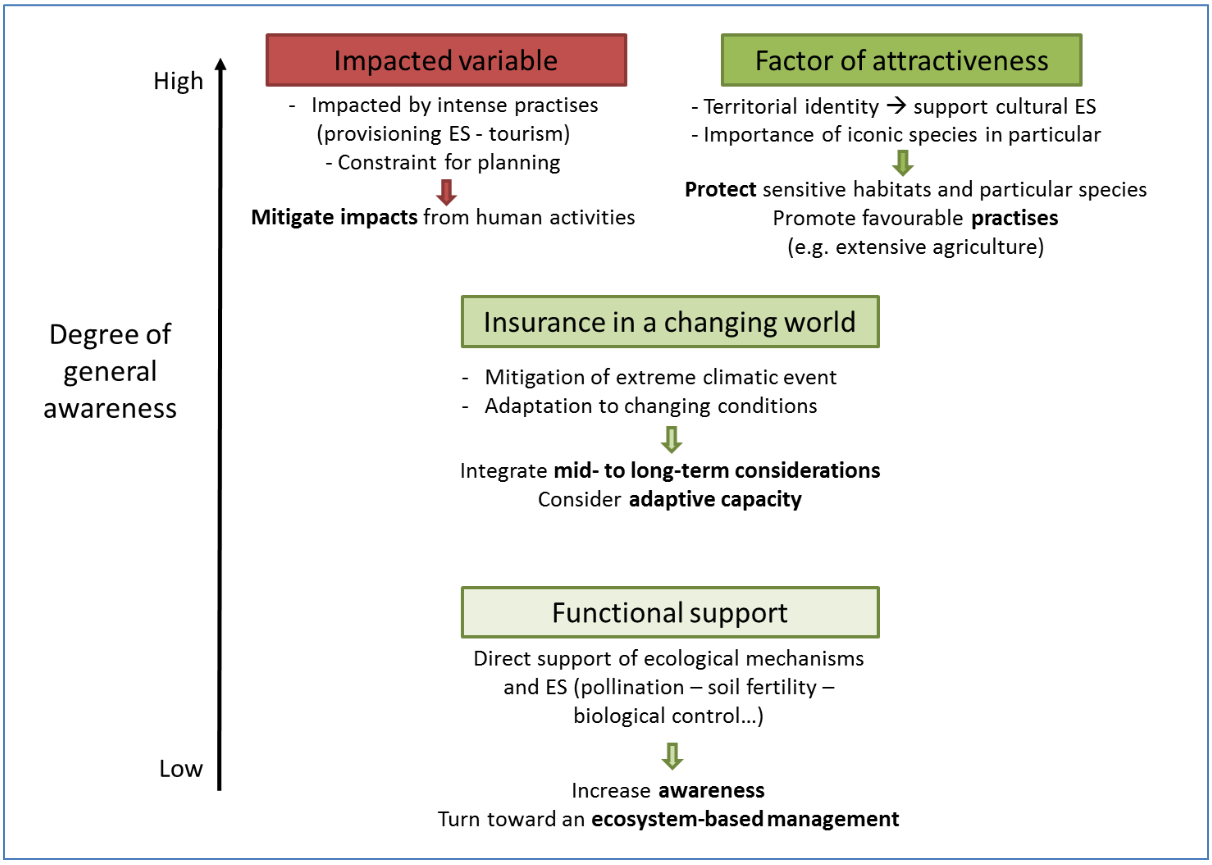


Figure 3: Four stances exploring the ways through which society conceives its interaction with biodiversity in the social-ecological system of the French Alps, ranked according to the general awareness conveyed by the stakeholders we consulted.

The four stances proposed (Figure 3) were unevenly mentioned by the stakeholders we consulted. In particular, the insurance value of biodiversity and its supporting role for functioning ecosystems were under-considered compared to biodiversity as the impacted variable or a factor of attractiveness. Increasingly considering the importance of biodiversity to sustain ecosystem functions and further ES supply hence appears as a challenge for scientists, managers and decision-makers. This holds true in the context of a changing world and 'simply' also under the current dynamic conditions (i.e. even without extreme severe changes, a high biodiversity supports higher opportunities for rich bundles of ES and sustainable interactions with ecosystems - Cardinale et al. 2012). Awareness-building efforts are thus required to bridge the gap, in the French Alps and probably beyond also.

C. Multi-dimensional links between ES and biodiversity

I proposed to approach the alpine system through a multi-layered description of some of its important ES bundles and of social perceptions of biodiversity.

A key output from this description relates to the various interconnections among ES and biodiversity that appeared:

i) spatially, leading to the identification of congruent bundles of ecological parameters and to their relation to landscape features beyond land cover categories,
ii) socially, as distinct demands are to be considered to decide over natural resource management, leading to relations across stakeholder groups of concerns,

iii) politically, as sectoral instruments appeared insufficient to manage complex systems and as policy instruments can affect untargeted environmental components through numerous rebound effects.

Our results stress the necessity of this kind of 'social-ecological system'-based approach to advance toward a "more nuanced and comprehensive understanding of human-nature interactions within human-dominated environments", which is a step required to sustainably manage them (Reyers et al. 2013).

III. The '*what*', '*how*' and '*who gets what*' trio – A synthesis of my PhD

To conclude this manuscript, I propose to follow the three axes presented by Simeon (1976) for studying public policy but which appear equally relevant to assess the interests and limits of what has been achieved here. First, I address the 'what' of my PhD, i.e. its scope, by summarising which aspects we considered and which were left aside. Second, I consider the 'how', i.e. our means, to present how we proceeded and also how we could further progress. Third, I will conclude on the 'who gets what', defined by Simeon as the distributive dimension, to expand on potential effects and extensions of this work.

A. What aspects were considered? Which are not?

1. Value-domains investigated

By using the concepts of ES and of social-ecological system, I explored the "bijective relation between ecosystems and society" (Barnaud & Antona 2014). In particular, our assessment of the alpine system included two value-domains of the ES framework informing biophysical aspects and socio-cultural aspects (Martín-López et al. 2014). We used them jointly to assess ES three facets (quantitatively and/or qualitatively), namely potential supply, demand and actual supply.

The value-domains selected to explore the ES framework "influence how the service in mind is characterized, which value dimensions are emphasized and how they are measured. More fundamentally, they influence which rationality is supported in the appraisal process" (Vatn 2009). In short, environmental assessment cannot be seen as a neutral process *uncovering* the values attached to nature; rather, it has been described as process *constructing* them (Vatn 2005, Gómez-Baggethun and Ruiz-Perez 2011). This is why the methods used to elicit values are called *value articulating institutions* (Jacobs 1997, in Vatn 2009): people will respond differently to the assessment they are proposed depending on the socio-institutional environment in which they express them (Kallis et al. 2013). In this perspective, epistemic commitments from the scientists designing environmental assessments can be expressed through the different weights and relevance given to distinct value domains explored and to the institutions chosen to articulate them.

One value domain that I did not consider for the alpine system assessment is the economic perspective, for at least two reasons. First, economic valuation is demanding in terms of methods, time and interpretation (Bateman et al. 2010). Adequate knowledge was lacking to carry it out in the context of the CONNECT French Alps case study, which made its implementation impossible. Second, it is my personal opinion that no added value would have been given to our understanding of the alpine system through an economic valuation of its ES. I share the concerns linked to ES economic valuation in general, and monetary valuation in particular, that consider such valuation exercises as early stages for the commodification of nature (e.g. Gómez-Baggethun et al. 2010, Gómez-Baggethun et al. 2011, Maris 2014). It has been demonstrated that, in the current state of the art, economic valuation is positively biased toward market-based ES at the expense of ES valued for alternative socio-cultural motivations and also more generally at the expense of regulating ES (Martín-López et al. 2014). I am not convinced that economic valuation could overcome this bias, whatever methodological progress is made to "capture *all* of the information pertinent to any particular environmental choice" (the *sufficiency claim* described by Vatn & Rombley 1994). Indeed, economic valuation relies on a hypothesis of *strong commensurability* involving comparability – i.e. that "there exists a single comparative measuring unit by which all different values can be ranked" (Martín-López et al. 2009). Assessing the total economic value of an ES would make possible to rank, substitute or compensate it (Luck et al. 2012). And yet, elements in the relations between human and nature might be beyond transferability, compensability and even commensurability (e.g. the uniqueness of relational values described by Murana et al. 2011 in Luck et al. 2012, see also Hauck et al. 2012, Jax et al. 2013). I nevertheless acknowledge that economic valuation could be useful in particular for studies that use monetary values to compare between management options (Boeraeve et al. 2014) and granted that they are carried out under specific conditions that i) ensure environmental additionality, ii) promote social equality, iii) avoid complexity blinding and iv) oppose enclosure of the commons (as developed by Kallis et al. 2013). Yet, I remain circumspect regarding its generalised use in environmental assessment as a prime (or even sole) driver for decision making.

2. A social-ecological system as a holarchy – Discussing scales

Our assessment focused on a regional scale (corresponding to NUTS-2 standards). This choice appeared coherent considering that the French Alps can be considered as a social-ecological system as such. The area has a biophysical coherence (the mountain massif), is acknowledged by governance (e.g. Massif committee – Alpine Convention) and is culturally identified by its inhabitants and by people beyond its borders (e.g. Alparc: The Alps – A unique cultural heritage http://www.alparc.org/the-alps/a-unique-cultural-heritage). The bundles of ES currently supplied are the result of historical interactions among alpine societies and the biophysical setting (Crouzat et al. in review). Our results were built to make sense at this regional scale only, with three main consequences. First, local interpretation of our quantitative results would not be relevant. Second, any sub-regional assessment should account for finer socio-cultural and biophysical specificities that we were not able to fully consider. Third, our results might only be of generic value for biophysically and socially comparable regions.

Figure 4 replaces our scale of concern among the nested components of a conceptual social-ecological system (Martín-López et al. 2009). The system is presented as a nested hierarchical system, or *holarchy*, where each layer is called a *holon* and has a dual nature, both as a whole and a part of the whole (Koestler 1978, in Waltner-Toews et al. 2002). Our study targeted intermediate layers, from ecoregion down to landscape levels in the ecological system and from national down to sub-regional institutions in the social system.

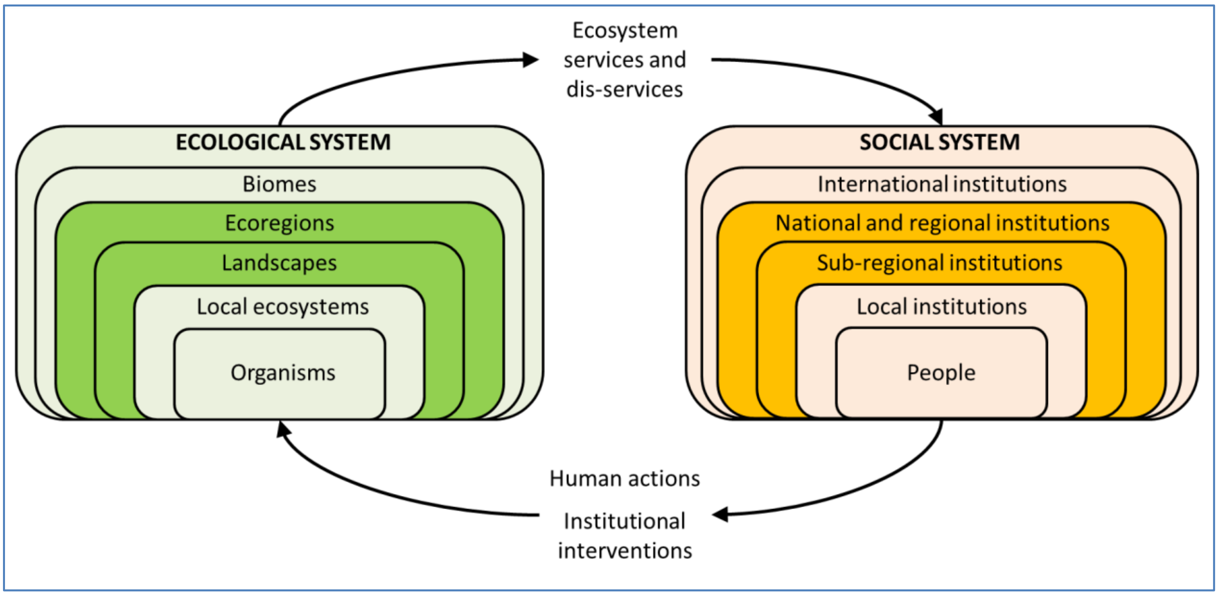


Figure 4: Scales of focus during my PhD (in dark backgrounds) replaced among the nested scales composing a conceptual social-ecological system (in light background). Each level can be considered a whole and a part of the whole, i.e. a 'holon', the overall system being called a 'holarchy'. Adapted from Martín-López et al. 2009.

The interest of the holarchy concept is to highlight the need for multi-scale analyses. Indeed, these would enable the discovery of 'emergent' properties that can be detected at a given level but arise from influences of upper or lower holons. A further improvement of my work would thus be to consider multiple holon levels. I use the term 'holon' rather than 'scale' to highlight that the lower and upper layers considered are also 'wholes' by themselves.

Regarding the ecological system

Regarding the ecological system, lower holons could be explored through more complex models able to account for finer functional properties (e.g. phenomenological or trait-based models). This would provide greater understanding of the functional links among ES and biodiversity (Díaz et al. 2007, Lavorel et al. 2013), which remained unexplored in this work. Due to a lack of data at fine resolution over regional geographical extents (species distributions – abiotic properties), such models were beyond my reach.

Combined with coarser data at higher levels, it would be interesting to analyse the emergent properties affecting ES supply through scales. We can assume that land cover configuration patterns through holons influence the supply of given ES linked for instance to hydrological flow regulation or species migration capacities. However, frameworks able to integrate properties of varying precision over scales remain scarce to date (but see Zaccarelli et al. 2008 for source/sink patterns of disturbance in the agricultural context of the Apulia region, Italy - even though not related to ES assessment directly).

Regarding the social system

Within the social system, considering lower holons could help distinguishing between stakeholder perceptions and concerns. While consultation results were presented in an aggregated fashion, the personal and institutional backgrounds of stakeholders have been proved to affect their perception and appraisal of ES (e.g. Lamarque et al. 2011, Castro et al. 2011, Lugnot & Martin 2013, Iniesta-Arandia et al. 2014). Our Influence Network Framework holds the potential to address such variations, which could be revealed by an extended stakeholder consultation paying attention to balancing their profiles and domains of concerns. Thus, different influence networks could be identified for different stakeholder groups. The next step of this work could be to explore the match between types of stakeholders according to their perception (i.e. groups of similar social representation) and rationales supporting policy orientations (i.e. the socio-political discourses). This has already been done for an alpine grassland landscape (Quétier et al. 2010) and results highlighted that social representations were unequally represented in the existing socio-political discourses identified at the European level. According the authors of this study, such a discrepancy can lead to the exclusion of the under-represented stakeholder groups from decision-making processes, possibly resulting on policy options of lower social acceptability.

Additionally, regarding governance, a multi-scale assessment would be the opportunity to further consider the articulation between high-level policies, regional scale planning schemes and local implementation of management strategies. In particular, I stress the interest of a focus at the municipal scale. Communes (and communities of communes) are the formal planning authority in France. However, our consultation highlighted a perceived lack of coherence and efficiency in land use and resource management especially in areas composed by many small independent municipalities, i.e. a lack of cooperation at supra-communal level. This may be partly due to economic lobbying on local decision-makers that was mentioned to threaten the sustainability of environmental management. Better understanding land planning determinants at local scale appears critical in the assessment of the alpine system, and further stakeholder consultation coupled with in-depth exploration of local governance instruments could help progressing in this direction.

B. How did we proceed? How could we further progress?

This assessment of the alpine system relied on the concept of ES used as a tool to describe the interactions among social and ecological spheres. I point out three key features from this concept to describe the potentials I see in using it to perform natural resource assessments, subjective to some further conditions or progresses.

1. Bundles to uncover complexity

ES have been criticised to blind complexity (Norgaard 2010). I believe this holds true mainly if ES are approached i) individually or in very restricted bundles, ii) through aggregated values, or iii) through unique value-domains, and in particular the economic domain. Challenging the idea that “single value outputs are what people understand” (Smith et al. 2011, Paracchini et al. 2011), the use of non-aggregated indicators, possibly describing multiple value-domains, seems essential to acknowledge complexity though the assessment of ES. Many examples exist that go in this way and demonstrate the suitability of ES to be considered individually and through alternative metrics to inform complex settings (as selected examples among others, Raudsepp-Hearne et al. 2010, Bryan et al. 2011, Castro et al. 2011, Bagstad et al. 2015, Crouzat et al. in review).

The challenge is rather to find a balance between ignoring complexity and overwhelming understanding by too much information. The ES concept used to assess bundles appears particularly relevant to address this challenge (Raudsepp-Hearne et al. 2010, Mouchet et al. 2014). The synthetic vision of the alpine system I proposed in the previous section is fundamentally based on alpine bundles of ES. Additionally, I believe bundles hold great potential for the study of rebound effects from policy decisions and management practises as they account for underlying spatial, ecological or social determinants beyond sectoral and land cover assessments (Bennett et al. 2009, Maestre et al. 2012). To detect bundles, I support methods of clustering as Self-Organising Map algorithms that display bundles of ES characteristic of areas with consistent social and biophysical backgrounds. From my experience, they enhance understanding and communication in a simple and yet integrative way. Several methods of clustering exist (e.g. hierarchical cluster analysis – principal component analysis) and present complementarities and differences. However, a clear methodology guiding their choice is still lacking to date and would be interestingly explored to enhance the consideration of bundles in ES assessments.

2. Transdisciplinarity to produce boundary objects

Abson et al. (2014) demonstrated that "the complexities discussed in the different ecosystem services research foci have not yet been integrated into a shared understanding or operationalization of the concept". In particular, they explored the conceptual keywords characterising distinct clusters of publications on the ES domain and showed a high compartmentalisation of research. Here, it is the potential of ES for being a *boundary object* that is being questioned. Boundary objects were defined as i) intersecting social worlds (e.g. across scientific disciplines / academic – non-academic partners), ii) plastic enough to be adopted by the different parties involved, and iii) robust enough to maintain a common identity across sites and partners (Star & Griesemer 1989). In short, such objects are a 'mean of translation' essential to develop and maintain understanding between distinct stakeholders (including scientists and decision-makers) collaborating on a common task (Castella et al. 2014). However, Barnaud & Antona (2014) pointed out the numerous debates that are linked to the ES concept (see also Chapter IV). They asked whether these would rather drive the use of ES toward 'dialogues of the deaf' than toward actual translation among stakeholders. Acknowledging the difficulties of handling such a 'hot' concept, they nevertheless concluded on ES as an "opportunity to increase dialogue and mutual understanding among people and disciplines". Indeed, ES concern academics from ecological, social or political sciences, as well as all citizens and decision-makers. Their definition is simple and broad enough to be understood by all and adapted to different settings and objectives. They have been used to co-produce knowledge from various sources (including local ecological knowledge) that can be further used to facilitate the mediation process between multiple stakeholders (for actual implementation, see for instance Palomo et al. 2011, Lamarque et al. 2014). To strengthen their status of boundary objects, ES science should foster the actual collaboration between disciplines in a *transdisciplinary* way, i.e. such that social and ecological approaches should actually "become enriched and empowered by an understanding and appreciation of alternative epistemologies" (Reyers et al. 2010). In the work presented in this manuscript, I tried to consider ES as boundary objects, although my work has been characterised by an academic vision *with intention of* policy and social relevance.

3. Integrative frameworks to inform multiple types of knowledge

Sustainability is as an objective often referred to for the use of the ES concept (MEA 2005, Mainka et al. 2005). The contribution of science to sustainability has been described as relying on three types of knowledge (ProClim - Forum for Climate and Global 1997): i) *systems knowledge*, proposing a descriptive understanding of a social-ecological system and of its current and potential ES, ii) *normative knowledge*, describing the targeted system states, and iii) *transformative knowledge*, required to shape and implement the transition from the existing to the target situation. ES assessments have been proved to generally favour systems knowledge upon normative and transformative knowledge (Abson et al. 2014), which raises some concerns as ES cannot be conceived as a neutral concept (Fisher & Brown 2014).

If some authors have highlighted the importance of considering ethical issues (e.g. Jax et al. 2013, see also Chapter IV), "few publications on ES engage deeply with normative issues" (Abson et al. 2014). Making explicit the values and judgments on 'what is desired or what is a good system state' should concentrate more efforts for ES assessment to become socially relevant. This is true as well for my work as I conceived it during my PhD project and I would like to pay further attention to this point during future projects. The kind of reflexive assessment I proposed in Chapter IV as a sequence of 8 questions could be interestingly reinforced and complemented to propose a normative framework broadly applicable to various ES assessments.

Further, to increase the ability of ES as a 'transformative tool for sustainability', there is a need for methodologies to consider governance aspects (both formal and informal) as well as social behaviours (motivations, communication, education), which remains pretty rare to date (Abson et al. 2014). Among promising methodologies, I see the interest of participative scenario (Lamarque et al. 2013), participative mental models (Moreno et al. 2014), fuzzy cognitive maps (Kok 2009), bayesian belief networks (Landuyt et al. 2013), social network analysis (Hicks et al. 2013) and influence networks (Crouzat et al. submitted). As mentioned in Chapter III, I believe that by including the three types of knowledge (namely systems knowledge, normative knowledge and transformative knowledge), ES could initiate triple-loop learning (Pahl-Wostl 2009) and thus favour adaptive (co-)management of natural resources (Armitage et al. 2008, Daily & Matson 2008). Although I could not propose a proper thinking on social learning and adaptive capacity during my PhD, I believe they are concepts of upmost importance to approach a social-ecological system and the methods listed here could help addressing them.

Overall, dealing with human-environment interactions remains challenging and no single method has been proven comprehensive enough to reach their complete understanding (Young et al. 2006). Rather, it is in the combination of several methods that the 'jigsaw puzzle' can be addressed. Analyses of different types (e.g. statistical analysis, discourse analysis and meta-analysis of case studies) can relevantly complement each other in scope. Additionally, using such a 'portfolio approach' (Young et al. 2006) is proposed as an interesting way to reduce the uncertainty still characteristic of numerous ES assessments (Seppelt et al. 2011, UK NEA 2011, Schulp et al. 2014).

C. Who gets what? What insights for following projects?

I have presented the results, interests and potential improvements for our bundle analysis in the French Alps region. To conclude, I turn to the road ahead and propose some milestones that I believe important to undertake in the context of the ICARE project.

Presented more in details in Chapter IV, ICARE is a collaborative action research project of restricted scope (i.e. a pilot project) focused on the territory of one community of communes close by Mont-Blanc area (*2CCAM 'Cluses Arve et Montagnes'*). Our common objective is to inform an environmental assessment using the concepts and methodologies I tested during my PhD, with the underlying commitment to support the consideration of (semi-)natural and agricultural area in future land planning. Collaboration with local authorities is required for the project to carry on, which should further include other stakeholders concerned with the issue. Whilst the project is still in its very first steps, it could benefit from some insights gathered throughout my PhD that I present below (Figure 5). In no case should this list be considered as exhaustive or prescriptive, as by essence the project is to be co-constructed.

Rather, it focuses on some elements that concern LECA's participation and for which I believe that the work previously carried out can provide relevant elements.

Elements concerning LECA's participation are structured around four main steps (Figure 5): i) framing the project, ii) defining supporting concepts and methods, iii) carrying out the assessment and iv) communicating results.

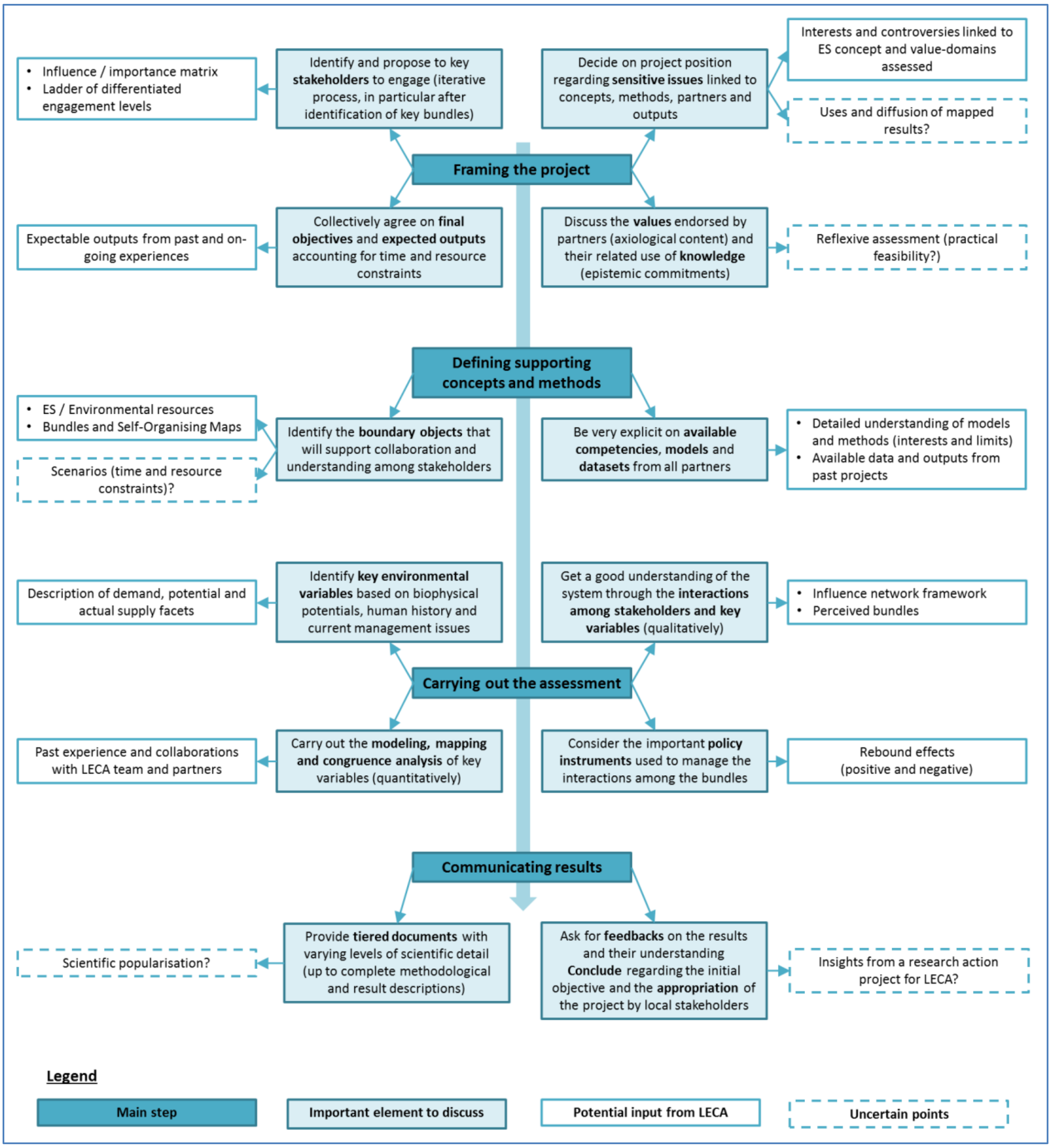


Figure 5: Contribution of LECA past experiences and knowledge to some important elements of the collaborative ICARE project.

Overall, the ICARE project will benefit from past and on-going experiences from various projects carried out by LECA team (VITAL, CONNECT, ESNET). They will provide experience on **concepts**, such as ES, ES facets, bundles or rebound effects. They will also be essential at a **methodological** level, quantitatively through inputs such as modelling ability, spatial congruence analysis or self-organising mapping, and also qualitatively via the frameworks we can propose to formalise a common understanding of the social-ecological system. Finally, LECA will also provide **practical inputs** such as data and models.

I stress the necessity to be very explicit on commitments and expectable outputs, so as to favour a transparent dialogue among partners. How to present our results so as to ensure that they are not over-interpreted (e.g. by an excessive zoom in regarding the input data resolution) remains a point of attention that I will thoroughly discuss with the project partners. We will also collectively agree on the level and means of scientific popularisation that should be reached as well as the stakeholder groups targeted beyond those explicitly engaged in the project.

Due to the restricted timespan and resources of the ICARE pilot project, a complete exploration of the social-ecological system, even though of restricted spatial extent, is beyond our reach. Thus, all steps described in Figure 5 will not be addressed and project partners will have to decide on which aspects focusing their contributions. If the biophysical assessment of the area will be led by LECA, the additional inputs we could provide remain subjected to further discussions and practical modalities as the project will carry on.

What the ICARE project will deliver in terms of final outcomes is still unknown. I hope that it will provide opportunities for exciting action research, “research in which the researcher has to allow the situation to take him/her where it will, research whose focus is in the change process itself” (Chekland 1985, in Castella et al. 2014).

D. Conclusion

To echo the first lines of this manuscript, in my PhD project I addressed ecological and social interactions in the French Alps cultural landscapes. Throughout my work, impacts of environmental management appeared critical both for the conservation of biodiversity and for the sustained supply of ES, further putting at stake human well-being. My results highlighted that modalities of environmental management affect both ES and biodiversity in multiple and differentiated ways, in particular depending on i) the intensity of practises used to benefit from ES of provisioning and cultural (nature tourism more specifically) categories and ii) land allocation choices. The determinants of environmental management were found to relate both to socio-cultural and biophysical aspects, for instance contrasted social demands for certain ES or guidance from policy and topographic or climatic constraints on ecosystem functions respectively.

To encompass the interrelated and dynamic influences that shaped landscapes through time, I followed a ‘social-ecological system’-based approach (Reyers et al. 2013). Overall, I considered biophysical and socio-cultural aspects at a conceptual level by working with specific objects (e.g. bundles of ES, formal governance institutions) and methods (e.g. self-organising maps, influence networks), and also at a pragmatic level through their application to the assessment of the French Alps system. In particular, this was achieved by i) exploring quantitative modelling and mapping methods for a pattern-based approach of multifunctionality, ii) proposing an innovating integrative framework to qualitatively describe social and ecological influence relationships, and iii) testing an extended approach of policy mix analyses through a collaboration with CONNECT partners. Normative aspects, including epistemic commitments, were explored at a conceptual level mostly, which calls for further experiences to get them more pragmatically applied.

My PhD project was conducted as an interdisciplinary approach of the French Alps social-ecological system with an intention of policy and social relevance. I greatly benefitted from the rich conceptual and methodological background of my fellow research partners. The next step I see for my work is to progress toward a transdisciplinary approach that would more fully endorse “values and norms of both society and science” (Reyers et al. 2010) while producing a comprehensive knowledge though collaborations across disciplines and spheres.

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Appendices

I. Appendices from Chapter I

A. Mouchet et al. (2014) An interdisciplinary methodological guide for quantifying associations between ecosystem services

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An interdisciplinary methodological guide for quantifying associations between ecosystem services

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ABSTRACT

Considering the increasing uptake of the concept of "ecosystem services" in landscape management and environmental policies, it is urgent to establish a consensual framework to assess the complex relationships among ecosystem services, considering both the supply- and the demand-sides. A diversity of approaches have been proposed to evaluate ecosystem services associations, but not all methods are equivalent and methodological choices need to be made depending on the scientific and policy questions at hand, as well as the type of data available.

Based on previous classifications of ecosystem service associations, we propose to characterize three broad types of associations considering the ecological (supply side) and socio-economical (demand side) aspects of ecosystem services: supply–supply, supply–demand and demand–demand. We then review quantitative methods available and propose guidelines to assess those three categories of relationships among ecosystem services and identify their explanatory variables following three steps: (i) detecting ecosystem services associations, (ii) defining bundles and (iii) identifying the explanatory variables of ecosystem services associations. For each step, strengths and weaknesses of different statistical analysis and machine learning methods are described.

The proposed interdisciplinary methodological approach takes one step toward embracing such complexity of socio-ecological systems as it considers ecosystem services delivery (supply–supply), stakeholders' needs (demand–demand), and on how stakeholders can benefit from the ecosystem services delivery (supply–demand). We illustrate how such a diverse spectrum of methods may apply for land management.

Highlights

- Characterization of three types of ecosystem services associations based on the ecological (supply) and socio-economical (demand) aspects of ecosystem services.
- Proposition of guidelines and methods to assess relationships among ecosystem services and identify their explanatory variables.
- Illustration of how a diverse spectrum of methods may apply in a context of

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1. Introduction

Facing an increasing anthropogenic pressure, ecosystems have been tremendously altered to the point of threatening the services they provide to society (Balmford and Bond, 2005, MA, 2005 and Swallow et al., 2009). The last decade has seen increasing efforts to incorporate sustainability of ecosystem service (ES) provision into policies and land management objectives (TEEB, 2010 and Perrings et al., 2011). However such an ambitious goal is challenged by the scarcity of knowledge on the consequences of specific environmental policies or management decisions for different ES and their associations, whether these policies or decisions target a single or several ES (DeFries et al., 2004 and de Groot et al., 2010). As an example, much of the recent focus on climate change mitigation through carbon sequestration has raised concerns on unintended consequences on biodiversity conservation and on other ES, even if some secondary benefits can be expected (Díaz et al., 2009).

Two mechanisms may lead to associations among ES: (i) the supply of several ES relies on the same ecosystem process, as in the case of wetlands acting as a buffer against climatic variability, providing flood control and shoreline stability; and, (ii) a given external factor may affect several ES at the same time as with the use of fertilizers positively influencing crop yield but decreasing water quality (Bennett et al., 2009). In the first case, the capacity of ecosystems to provide a variety of ES, i.e. multifunctionality, stems from linkages among basic ecosystem processes through organismic trade-offs (Lavorel and Grigulis, 2012). In the second case, the way one service is managed will likely affect one or more other ES. As a result of these associations, some ES co-vary positively, for instance biological control of pests may improve food supply by limiting crop damage, while some ES may co-vary negatively, as for food supply degrading water quality through the use of fertilizers. Ecosystem management strategies aiming at maintaining or enhancing the supply of a given ES need to account for such basic linkages to enhance the supply of several interrelated ES (Rodriguez et al., 2006, Bennett et al., 2009 and Carpenter et al., 2009).

Recent studies have taken two different approaches to assess ES associations: the evaluation of associations at a given location and time versus the evaluation of associations across sites and/or through time. In the first case, the assessment is a static snapshot of ES associations and is insufficient to conclude that observed associations between ES can be generalizable to a larger extent. The second case relates to what Raudsepp-Hearne et al. (2010) call a “bundle” that is to say “sets of ES that appear together repeatedly”. Although conceptually divergent, both approaches have been presented as “trade-off assessments”. Besides, “trade-off” has been equally applied to ecological relationships between ES (Egoh et al., 2008) and to the congruence between ES demand and ES supply (García-Nieto et al., 2013). Ecological trade-offs underpinning ES supply are the heart of all types of trade-offs and should be properly assessed to efficiently anticipate demand–supply congruencies and the cost–benefit balance for the management of multiple ES (Seppelt et al., 2011).

Given such a lack of consensus on definition and approaches, the aim of this paper is to review and streamline terminology for ES “trade-offs” (see also Box 1), and then to synthesize state-of-the-art knowledge in order to propose methodological steps and techniques for assessing different types of associations between ES depending on their nature and on research objectives. Besides, identifying those environmental or social pressures linked with ES associations is a key step, although usually overlooked, essential to manage for bundles of ES and predict their dynamics in time and under alternative policies (Nelson et al., 2009 and Power, 2010). To address this gap, we incorporate an overview of methods to identify explanatory variables of ES associations, a first step toward the analysis of associated mechanisms. We conclude by considering key elements that should be taken into account when analyzing ES associations with the objective of informing land management and policy development.

Box 1. Definition of some the main concepts discussed in this article

Ecosystem service (ES) has been previously defined as "the benefits provided by ecosystems that contribute to making human life both possible and worth living" (Díaz et al., 2006) or "the contributions that ecosystems make to human well-being, and arise from the interaction of biotic and abiotic processes" (Haines-Young and Potschin, 2010). Díaz et al. (2006) further argued that "ecosystem services are context-dependent; that is, the same ecosystem process can produce an ecosystem service that is highly valued by one society or stakeholder group but not highly valued by other societies or groups." In that sense, ecosystem services are defined according to beneficiaries. Ecosystem services Villamagna et al. (2013) and Schröter et al. (2014) distinguished two aspects in a service: capacity and flow. ES capacity is "the long-term potential of ecosystems to provide services appreciated by humans in a sustainable way, under the current management of the ecosystem. Capacity may be increased or decreased over time through ecosystem management and land use conversion." (Schröter et al., 2014 and references cited). ES capacity also refers as the potential of an ecosystem "to deliver services based on biophysical properties, social conditions, and ecological functions" (Villamagna et al., 2013 and references therein). ES flow is "the actual use of ecosystem services and occurs at the location where an ecosystem service enters either a utility function […] or a production function […]" (Schröter et al., 2014) and is also "the service actually received by people, which can be measured directly as the amount of a service delivered, or indirectly as the number of beneficiaries served" (Villamagna et al., 2013). However, ES flow is not ES demand.

ES demand is "the amount of a service required or desired by society" (Villamagna et al., 2013). For that reason, the demand of a given ES may exceed the capacity of an ecosystem to deliver the service.

ES supply represents to the capacity of the structures and processes of a particular ecosystem to provide a specific bundle of ecosystem services within a given time period (modified from Harrington et al., 2010 and Burkhard et al., 2012). In this paper, we consider that "ES supply", "ES delivery" and "ES provision" are synonymous terms.

ES bundle refers to a "sets of ES that appear together repeatedly" (Raudsepp-Hearne et al., 2010). In a bundle, ES can be positively (synergy) or negatively (trade-off) associated. The associations can rise from common underpinning processes or as a response to common pressures (Bennett et al., 2009, but see the main text for further details).

2. Streamlining classifications of ecosystem services associations

The use of "trade-off" as a generic term for ES associations (in TEEB, 2010 for instance) may be misleading. "Trade-off" applies when two entities (here ES) show opposing trends (i.e. when the level of one ES supply increases, the level of the other ES decreases). When the supply of two ES co-vary positively, "synergy" would be more appropriate (already used in Bennett et al., 2009, Egoh et al., 2009, Lavorel et al., 2011 and Haase et al., 2012). However, in the assessment of relationships among ES, one must first distinguish the static associations (positive or negative) between ES from associations robust in space, and potentially long-lasting, although the strength of association may fluctuate. The term "association" should prevail over “trade-off”, “compromise” or “synergy” when the assessment of ES relationships is just a snapshot. If the repeatability criterion given by Raudsepp-Hearne et al. (2010) is met then one can use the terms “bundle” of “trade-off” or “synergy” instead of “association”. In the literature, “trade-off” has also been used to name various types of compromises: ecological compromises between ES (e.g. Vihervaara et al., 2010), a temporal trade-off in the supply of an ecosystem service (e.g. Koch et al., 2009), management compromises between ES (e.g. White et al., 2012), compromises between ES supply and demand (e.g. Kroll et al., 2012), compromises between cost and benefit (e.g. Viglizzo and Frank, 2006), and compromises between different beneficiaries (e.g. Martín-López et al., 2012). Two broad classifications of trade-offs have been proposed in the literature. In the first classification established by Rodriguez et al. (2006) as part of the Millennium Ecosystem Assessment (MA), ES associations, or so-called “trade-offs” in their framework, were classified into four categories: (i) spatial trade-off, the spatial lag between ES production and the delivery of this or other ES; (ii) temporal trade-off, the temporal lag in the ES delivery resulting from management decision or natural processes; (iii) reversible trade-off, the ability of a ES to return to its initial supply after a disturbance in the production of the given service in relation with the resilience of underlying natural processes; and (iv) trade-off among services, the positive or negative effects of the supply of one ES on the supply of other ES. The Economics of Ecosystems and Biodiversity (TEEB) assessment (2010) proposed a classification with a partly similar terminology, but some different definitions, stated as: (i) spatial trade-off, the spatial lag between the benefit and the cost related to the targeted ES; (ii) temporal trade-off, the time lag between the benefit of a service and the associated cost because the deterioration of this or other ES in the future; (iii) trade-off between beneficiaries, where beneficiaries can be either “losers” or “winners” depending on who bears the cost of or the benefit of the ES supply; and (iv) trade-off among ES, addressing management of one ES at the expense of another. While the MA classification (Rodriguez et al., 2006) focuses on the consequences of ecological trade-offs for ES supply, TEEB's is framed in terms of economic benefits and costs for ES demand (except for the last category). Currently, these two typologies of relations between services, ecological versus socio-economic or supply versus demand, coexist in the literature under the generic term of “trade-off”.

In order to guide the quantitative assessment of ES associations, we propose to streamline previous typologies and thereby reconcile previous classifications, by accounting for both ecological (i.e. supply) and socio-economic (i.e. demand) aspects of ES associations. Hereafter, “ES associations” will refer to both punctual associations or associations repeated in time and space. This would yield three possible combinations (Fig. 1): (i) supply–supply, referring to trade-offs and synergies in simultaneously provided ES; (ii) supply–demand, to describe the spatial or temporal lag between ES supply and social benefits; and (iii) demand–demand, referring to the arbitration between different and divergent stakeholders’ interests. Table 1 summarizes the main objectives and characteristics of each of these combinations.

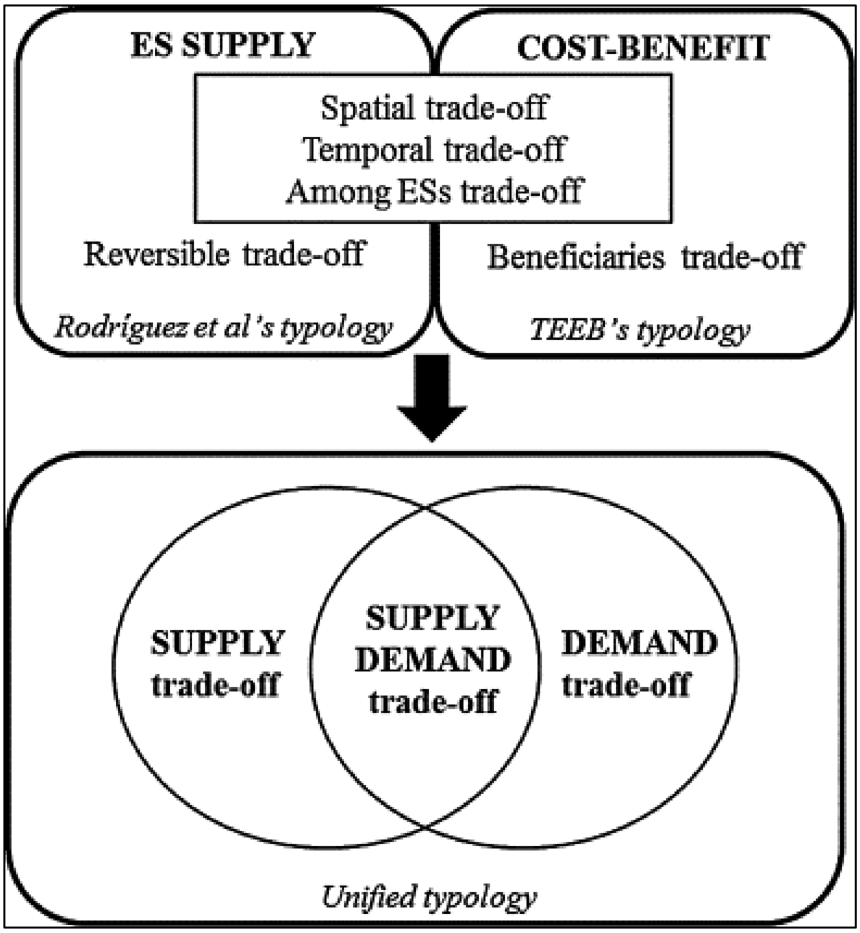

Figure 1: The unified typology of ecosystem services trade-offs. This classification seeks to merge both Rodríguez et al. (2006) and TEEB (2010) frameworks. Here "trade-off" encapsulates both trade-off and synergy.

Table 1: Characterization of the three broad types of ES associations identified on the basis of their ecological and socio-economic aspects and illustrated by a selection of key questions and applications.

	Supply–supply case	Supply–demand case	Demand–demand case
Main challenge	To explore the spatial congruency in ES supply in order to design "win–win" management and policies supporting multifunctionality and reconciling nature protection and ES delivery	To analyze the spatial or temporal mismatch between ES supply and the derived social benefits	To explore the different stakeholders' interests regarding the use and demand of ES
Associated research questions	(1) To what extent and why does the supply of one ES correlate or overlap with other ES or with biodiversity?	(1) How well do the supply of ES and their use, or valuation by beneficiaries, spatially match?	(1) To what extent do ES demands by different stakeholders concur or conflict?
	(2) Where are areas of high and low supply of multiple ES (i.e. hotspots and coldspots, respectively)?	(2) Is there a temporal mismatch between the ecological processes behind ES supply and its use by beneficiaries?	(2) How do stakeholders economic or social status influence trade-offs among their ES demands?
	(3) How is the distribution of ES bundles influenced by land management and/or by the distribution of biodiversity?		
Examples of application	(1) Identification of places where simultaneously conserving biodiversity and delivering a diverse flow of ES (e.g. Chan et al., 2006, Egoh et al., 2009 and Bai et al., 2011)	(1) Identification of places where simultaneously conserving biodiversity and delivering a diverse flow of ES (e.g. Chan et al., 2006, Egoh et al., 2009 and Bai et al., 2011)	(1) Identification of places where simultaneously conserving biodiversity and delivering a diverse flow of ES (e.g. Chan et al., 2006, Egoh et al., 2009 and Bai et al., 2011)
	(1) A spatial scale mismatch has been found between the demand for and the supply of energy, food and water services along a rural-urban gradient in the Leipzig-Halle region (Germany) (Haase et al., 2012 and Kroll et al., 2012)	(1) A spatial scale mismatch has been found between the demand for and the supply of energy, food and water services along a rural-urban gradient in the Leipzig-Halle region (Germany) (Haase et al., 2012 and Kroll et al., 2012)	(1) A spatial scale mismatch has been found between the demand for and the supply of energy, food and water services along a rural-urban gradient in the Leipzig-Halle region (Germany) (Haase et al., 2012 and Kroll et al., 2012)

3. Developing a methodological framework for quantifying ecosystem service associations

Identifying and quantifying the associations between ES is essential to foresee the impact of environmental changes and management on ES supply and thus on ES beneficiaries, as well as to understand how management choices promote trade-offs or synergies for a specific ES or shape the composition of bundles of ES. Using recent methodological advances that have been mainly applied in ecology so far, we propose to investigate the associations among ES following three successive steps: (i) detecting ES associations, (ii) identifying bundles of ES and, (iii) exploring potential drivers (Fig. 2 and Table 2 for a summary of all methods described below).

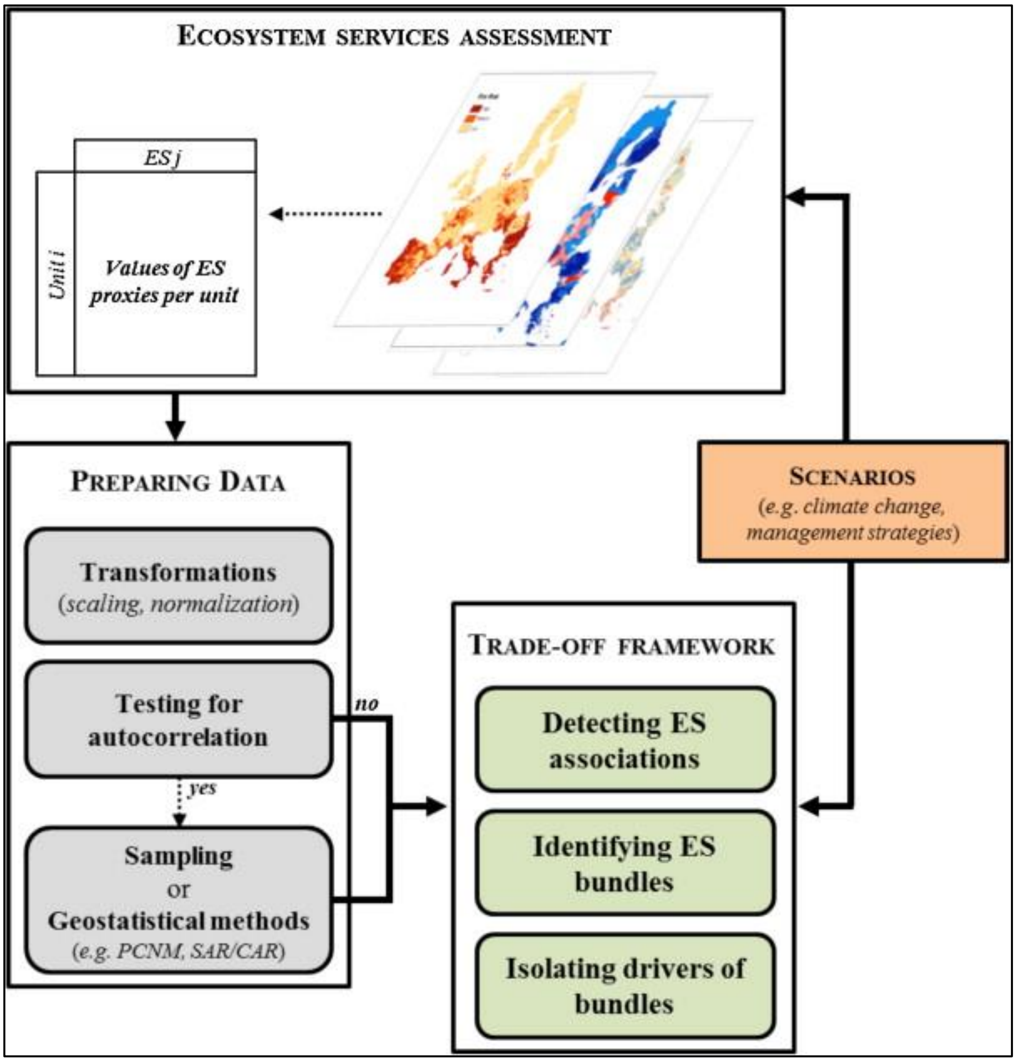

Figure 2: Illustration of the methodological framework for assessing trade-offs. ES indicators may be measured in the field (for either ecological or socio-economic data) or modeled from scenarios and then mapped or directly expressed as ES values per unit (i.e. sites or time steps). ES data may be transformed and normalized to fit validity conditions of statistical methods. See Supplementary material for more information

Table 2 (next pages): Overview of the quantitative methods available for analyzing ES associations. Methods presented in the table may apply to more than one category of ES association (i.e. "supply–supply", "supply–demand", "demand–demand"). Several methods that are mainly dedicated to visualization of ES associations (e.g. star diagram, network analysis) are also mentioned in the main text but not in this table.

Step of the frame work	Method	Number of ESs	Dimension and types of the variables	Including spatial and temporal variability	Further details	Examples of hypotheses to be tested	References
Detecting defining bundles	**Association coefficient**						
	Correlation coefficients	Two	Quantitative variables	See Dutilleul. (1993) for a modified t-test accounting for spatial autocorrelation Temporal correlation should be tested using time-series methods	Pearson's coefficient when normally distributed. Otherwise Spearman's coefficient	Which ES are associated?	Chan et al. (2006), Naidoo et al. (2008), Anderson et al. (2009), Egoh et al., 2008 and Egoh et al., 2009, Eigenbrod et al. (2010), Raudsepp-Hearne et al., (2010), Smart et al. (2010), Willemen et al. (2010), Bai et al. (2011), Butler et al. (2013), Gos and Lavorel (2012), Casalegno et al. (2013)
	Chi-squared test	Two	Categorical variables	Not spatially explicit Can be applied to the ES supply/demand at two different time steps	Both variables are represented in a contingency table	Hypotheses to be tested are similar from those tested using correlation coefficients	–
	Overlap analysis	Two and more	Quantitative and/or qualitative variables	Spatially explicit Can be used to estimate the temporal changes in ES associations	Requires a supply threshold to convert a qualitative ES index to a binary one	Are supply hotspots spatially congruent? Are demand and supply spatially congruent? Is there a temporal change in the spatial distribution of demand and/or supply?	Egoh et al., 2008 and Egoh et al., 2009, Swallow et al. (2009), Eigenbrod et al. (2010), O'Farrell et al. (2010), Bai et al. (2011), Gos and Lavorel (2012)
	Ordination						
	Principal Component Analysis (PCA)	Two and more	Quantitative variables	Not spatially explicit Can be diverted to included time steps instead of sites but time-series methods are more appropriate for this purpose	–	Which services are negatively or positively associated?	Raudsepp-Hearne et al. (2010), Smart et al. (2010), Maes et al. (2012)
	Multiple Correspondences Analysis (MCA)		Binary variables		Requires a supply threshold to convert a qualitative ES index to a binary one		–
	Factorial Analysis for Mixed Data (FAMD)		Quantitative and qualitative variables				–
	Clustering						
	K means	Two and more	Quantitative and/or qualitative	Spatially explicit when the *K* means or SOM are used to classify localities into groups which are then projected onto maps	*K* means can objectively classify ES into groups from the original data matrix or from the outputs of an ordination method. Usually, the number of groups is defined a priori in *K* means and SOM procedures. Both *K* means and SOM can help visualizing localities with similar combinations of ES supply or demand values	Which services are consistently associated? Which localities exhibit similar ES associations?	Raudsepp-Hearne et al. (2010)
	Self-Organizing Maps					Which localities have the same bundles?	–
	Overlap analysis	*See above*			–	Are associations spatially or temporally repeated?	*See above*

Appendix

Step of the frame work	Method	Number of ESs	Dimension and types of the variables	Including spatial and temporal variability	Further details	Examples of hypotheses to be tested	References
Identifying drivers of bundles	***Distance approach***						
	Mantel test and derivates	One or more response variable – one or more explanatory variables	Qualitative or quantitative response variables Quantitative and/or explanatory variables	Not spatially explicit Space and time might be included as an explanatory variable	The choice of the distance metric depends on the type of data (e.g. binary, mixed)	Is the similarity in the ES association between two localities explained by a similar combination of drivers?	–
	Analysis of Similarity (ANOSIM)				Non-parametric alternative to Mantel tests		–
	Raw data approach						
	Analysis of variance						
	ANOVA	One response variable – one or more explanatory variables	Quantitative response variable(s) Explanatory variables are categorical	Not spatially explicit Space and time might be included as an explanatory variable	The response variable should be an integrative index of multiple ES supply	Does the ES association vary through the drivers' states? Do ES association changes along a gradient of management strategies?	Willemen et al. (2010)
	MANOVA	Two or more response variable – one or more explanatory variables			When explanatory variables are quantitative and categorical, ANCOVA and MANCOVA are applied	–	–
	Co-inertia		Quantitative and/or qualitative variables		Co-inertia is a combination of ordination methods	How does the co-variation of drivers shape the covariation within a bundle?	–
	Regression-based model						
	Generalized Linear Model (GLM)	One response variable – one or more explanatory variables	Quantitative response variable Explanatory ones can be quantitative and/or qualitative	Not spatially explicit Space and time might be included as an explanatory variable	Models various type of relationships (e.g. Gaussian, log) and alows for prediction	Can the overall ES supply be explained by a set of environmental and/or socio-economic factors? Which is the most influential demand for ES on the overall ES supply? How will a bundle evolve with future changes in drivers?	Steffan-Dewenter et al. (2007), Smart et al. (2010), Fisher et al. (2011)
	Generalized Additive Model (GAM)				Models smoothed non-linear relationships (unlike GLM) and allows for prediction		–
	Autoregressive model			Spatially explicit	SAR and CAR require stationarity. When this condition is violated, a "moving window" method could be applied		–
	Machine-learning methods						
	Decision trees	One	Quantitative or	Not spatially explicit	Decision trees algorithms will	Can the overall ES supply be	–

Step of the frame work	Method	Number of ESs	Dimension and types of the variables	Including spatial and temporal variability	Further details	Examples of hypotheses to be tested	References
		response variable – one or more explanatory variables	qualitative response variables Quantitative and/or explanatory variables	Space and time might be included as an explanatory variable	produce either classification or regression trees whether the response variable is categorical or quantitative, respectively. Random Forest (RF) and Boosted Regression Trees (BRT) mainly differ in their way to split the dataset in groups. Both RF and BRT allow for predictions	explained by a set of environmental and/or socio-economic factors? Which is the most influential demand for ES on the overall ES supply? How will a bundle evolve with future changes in drivers?	
	Artificial Neural Networks (ANN)		–		May be hard to interpret	Which is the most influential demand for ES on the overall ES supply? How will a bundle evolve with future changes in drivers?	–
	Time-series methods						
	ARMA and derivates	One response variable – one or more explanatory variables	Quantitative variables	Not spatially explicit	–	Did the temporal changes in demand drive the changes in supply?	–
	VAR	Two or more response variable – one or more explanatory variables	Quantitative variables		–	To what extent the variability in several potential drivers influenced the temporal changes in ES associations?	–
	Canonical analysis						
	Canonical Correspondence Analysis (CCA)	Two or more response variable – one or more explanatory variables	Qualitative or quantitative response variables	Not spatially explicit Space and time might be included as an explanatory variable	CCA is used when the response variables are binary (0–1) or proportion	Basically, the same hypotheses as regression-based models. The difference is that the response variables can be ES proxies instead of a synthetic index so one can investigate which driver have the greatest impact on which ES	Lamarque et al. (2014)
	Redundancy Analysis (RDA)		Quantitative response variables Quantitative and/or explanatory variables		–		Smart et al. (2010)

The objective of this framework is to present a set of quantitative methods to assess ES associations within the three categories: supply–supply, supply–demand and the demand–demand trade-offs (or synergies). Most methods mentioned here can be applied to more than one of the categories of ES associations. ES associations can also be visualized using star (also known as radial, amoeba or flower) diagrams, bar charts, scatter plots, box plots or other types of plots depending on the nature of data and the outcome to capture. These visual methods will be only briefly discussed here.

3.1. Detecting ecosystem service associations

Once ES have been quantified, spatial or temporal trends in the distribution of two or more ES (indicator) values can be compared to find significant associations among ES.

The simplest approach to deduce positive and/or negative associations among ES is visual map comparison to outline spatial relationships (Anderson et al., 2009), trade-off curves to detect trends (e.g. Viglizzo and Frank, 2006 and White et al., 2012) or star diagrams to compare the relative provision of ES within a bundle (Foley et al., 2005 and Raudsepp-Hearne et al., 2010), but none of these graphic methods provide a quantification of the strength of the association. The most popular quantitative method to assess associations among continuous quantitative indicators is pairwise correlation coefficients (Table 2). In the case of two categorical indicators, a chi-square test on the two-way contingency table can replace the correlation analysis. However, multivariate analyses represent a better alternative when considering more than two ES and are flexible regarding the nature of the indicator (i.e. quantitative, qualitative): Principal Component Analysis (PCA) when all ES indicators are quantitative, Multiple Correspondences Analysis (MCA) when all ES indicators are qualitative (nominal or binary) and Factorial Analysis for Mixed Data (FAMD – which combines a PCA on quantitative variables and a MCA on qualitative ones) to handle a combination of quantitative and qualitative indicators simultaneously. Regression-based methods between two ES indicators can also detect ES associations, but their use goes beyond detection as regressions also imply directional causation (unlike correlations) and address the search for more mechanistic linkages among ES (Bennett et al., 2009). Still, regression-based models can get at causality only when the methodological framework has been set to test for causal relationships, that is to say essentially by using experimental systems and predictors directly assessing the underlying mechanisms.

However, none of the above methods is spatially explicit (except visual map comparison), although they can be performed with spatial data (see Table 2). Overlap analysis, and the related coincidence or congruence analyses, is a very simple and intuitive way to run a spatially explicit detection of associations. Basically, overlap analysis quantifies the percentage of cells where two ES are provided at the same time, with several possible implementation methods (Chan et al., 2006). In addition to the supply–supply case, this method may be particularly appropriate for simple detection of the other two types of associations – e.g. the spatial congruence between the demand for energy and energy production from biomass or hydropower (Kroll et al., 2012). This pairwise method may be extended to the identification of multiple ES associations with, for instance, the mapping of the "richness" in ES (i.e. number of ES) supplied at a given unit (e.g. a pixel, an administrative or ecological unit) (Smart et al., 2010 and Bai et al., 2011).

More recently, network analysis has been used for visualizing and quantifying the relations among ES on the basis of different stakeholders' perceptions. In practice, a network represents the interactions (links), either trade-offs or synergies, among ES (nodes) as prioritized by one stakeholder (Hicks et al., 2013). A comparison between stakeholders can be performed through the comparison of network diagrams resulting from each stakeholder's priorities. In network analysis, two measures are commonly used to quantify ES associations: degree centrality (i.e. the number of links connecting an ES to other ES) and betweenness centrality (i.e. the number of shortest pathways linking two ES, running through a third ES). Although this method is initially not spatially explicit, the comparison of network diagrams corresponding to different locations (e.g. municipalities) could help describing the spatial variations in trade-offs and synergies in a "demand–demand" context.

Lastly, temporal trends in ES supply have often been overlooked (but see Swallow et al., 2009, Lautenbach et al., 2011, Carreno et al., 2012, Haase et al., 2012 and Kroll et al., 2012). A very simple way to assess temporal associations is to quantify and to compare the percentage of change in an aggregated index of multiple ES associations between two periods. However, specific methods may be required to account for the temporal autocorrelation in ES supply (see "Dealing with autocorrelation" in the appendix). Cross-correlation measures the similarity of two time-series by expressing the linear correlation coefficient as a function of time lag (Legendre and Legendre, 1998). Besides, ecological and socio-economic processes underpinning ES associations may fluctuate periodically (e.g. seasonality). In that latter case, time-series analyses may help determine if the fluctuations in ES multiple supply or ES associations depart from regular variations.

3.2. Defining ES bundles

The previous set of methods only gives a static assessment (i.e. at one place and/or one time step) of associations among ES but ES associations should be consistent in space and, preferably in time as well, to be considered as bundles (Raudsepp-Hearne et al., 2010). First, cluster analyses can help to objectively define the groups of ES that are significantly associated. It is important to bear in mind that different cluster analyses can produce different clusters as a result of the hypotheses specific to each clustering algorithm. Hierarchical clustering has successfully been used to define ES bundles using the distance between the economic values (Martín-López et al., 2011) or social preferences (Martín-López et al., 2012). As an alternative, the K-means clustering algorithm can be applied to segregate ES into a pre-defined number of groups by minimizing within-group variability. Additional analyses can then be performed to obtain a more dynamic picture of ES associations by estimating their recurrence in space and time. A way to do so would be to compare correlation coefficients, multivariate or overlap analyses among different spatial units to check the spatial consistency of the observed associations. Self-Organizing Maps (Kohonen, 1990) should also help visualizing spatial clustering of services supply or demand. Temporally, ES associations may be inferred from the comparison of current ES supply to historical time series (Lautenbach et al., 2011) or to future scenarios (Nelson and Daily, 2010). To our knowledge, only few assessments of ES bundles applied clustering or repeatability analyses in spite of their simplicity (see references in Table 2).

3.3. Identifying drivers of ES associations

Critical progress in understanding the dynamics of ES bundles requires the identification of their potential drivers and causes (Bennett et al., 2009). Indeed, establishing the spatial (and temporal) congruence of several ES supply does not mean that ES arise from the same process(es). The types of questions that need addressing include: Do ES associations arise from one (or more) shared ecosystem process(es)? Are ES associations driven by social demand? Does landscape management influence the ES associations? To what extent does the way ES are modeled induce ES associations in assessments? In the following we outline available methods to explore the explanatory variables of ES bundles, whether they are ecological processes underpinning ES supply or socio-cultural factors influencing ES demand, and whether the associations of interest are supply–supply, supply–demand or demand–demand (Table 2). In this way, "explanatory variables" encapsulate both exogenous drivers (e.g. industrial production), causing environmental change in the socio-ecological system, and pressures (e.g. use of fertilizers) quantifying the effect of exogenous drivers on a given socio-ecosystem (Harrington et al., 2010).

ANOVA (ANalysis Of VAriance) is well suited to test whether a quantitative response variable, e.g. an aggregate index of ES supply, significantly varies between states of one or more explanatory variables. The extension of ANOVA to the case of a multivariate response variable, MANOVA (Multivariate ANalysis Of VAriance), would be more appropriate to the study of bundles (i.e. several ES indicators). Beyond these, co-inertia analysis is a more flexible multivariate method regarding variable types (quantitative and/or qualitative) and normality, which couples different methods (e.g. a PCA on quantitative ES indicators and a MCA on qualitative environmental variables) to maximize the co-inertia between, in this case, one table for ES values and one table of explanatory variables. Although this method has not yet been applied to identifying explanatory variables of ES bundles, it would be particularly appropriate to visualize how the co-variation of multiple explanatory variables (e.g. primary production, GDP) may shape the co-variation of several ES. However, only canonical analyses (i.e. Canonical Correspondence Analysis, CCA, and Redundancy Analysis, RDA) allow a quantitative test for causal relationships between a multivariate response variable (i.e. ES indicators) and explanatory variables. Canonical analyses, by combining ordination and multiple regressions, aim at finding the combination of explanatory variables that best explains the dispersion of ES values. For instance, García-Llorente et al. (2011) showed the relationships between those functional groups of aquatic plants underlying the ES delivery and the economic values assigned by stakeholders to these ES through performing a CCA. Finally, RDA has been commonly used for analyzing the socio-cultural explanatory variables of demand–demand trade-off or synergy (Hicks et al., 2009 and Martín-López et al., 2012). It is worth noting that the outcomes of canonical analyses may be biased by spatial autocorrelation (see “Dealing with autocorrelation” in the appendix) as well as the classical regression models presented below.

Alternatively, Mantel tests and distance-based methods (e.g. Multiple Regressions on distance Matrices, MRM, Congruence Among several Distance Matrices, CADM, ANalysis Of SIMilarity, ANOSIM), which use distance matrices as inputs for response and explanatory variables, may be applied to identify what drives differences in ES supply (e.g. among sites), rather than which variables influence bundles variability as done by raw-data approaches. However, distance methods should be used with care as they weakly detect complex relationships among matrices, underestimate the coefficient of determination of the variation explained by the spatial structure (Legendre and Fortin, 2010) and may not be valid when variables are autocorrelated (Guillot and Rousset, 2013). Raw-data approaches should be preferentially picked over distance methods unless the hypothesis is explicitly formulated in terms of distances.

Another strategy would consist in regression of the potential explanatory variables against the overall level of ES supply using an aggregated estimator of ES bundles. Such synthetic indices of ES supply have been published, including the “richness” in ES (Plieninger et al., 2013), the sum of standardized (Maes et al., 2012) or weighted (Gimona and van der Horst, 2007 and Kienast et al., 2009) ES values, or the evenness in ES supply calculated using the Simpson's index (Raudsepp-Hearne et al., 2010). It is worth noting that there is a conceptual difference between a multivariate approach of ES bundles and using an aggregated index. The first approach will relate ES co-occurrence or segregation within a bundle to the variability of one or more explanatory variables, whereas the second one will give insights into what drives multifunctionality. Using an aggregated estimator of ES supply entails two methodological issues: (i) defining a threshold of supply when calculating richness (see Appendix) and (ii) including qualitative estimators of ES supply. In this latter case, qualitative estimators should be removed from the analysis or transformed into a dummy (0–1) variable. Then relationships between multiple candidate explanatory variables and the aggregated estimator can be tested using Generalized Models (generalized linear models, i.e. GLMs, or generalized additive models, i.e. GAMs), depending on the linearity of responses and the complexity of response shapes. Given potential issues of spatial autocorrelation (see Appendix), the spatial regression methods SAR (Simultaneous AutoRegressive model) and CAR (Conditional AutoRegressive model), which have been specifically designed for this purpose, could be used, but they may be less efficient than GLM or GAM (see Appendix). In spite of their relative simplicity and currency in ecology, these methods have been rarely used in analyses of ES associations (but see Steffan-Dewenter et al., 2007 and Fisher et al., 2011).

On the contrary, different regression models are commonly performed in the stated-preferences economic valuations to identify socio-cultural explanatory variables determining the ES demand and the ES bundles. Stated-preference techniques (i.e. contingent valuation and choice modeling; Bateman et al., 2002) create hypothetical markets through questionnaires in order to estimate the economic value of different ES. On one hand, in the contingent valuation method, researchers directly ask people how much they would be willing to pay (or accept) for a change in the quantity or quality of one or more ES. On the other hand, choice modeling elicits social preferences by asking individuals to choose their preferred option from a series of alternatives of choice sets (with different scenarios), which are described in terms of different attributes associated with ES. Here, choice modeling employs the behavioral framework of random utility theory, in which it is assumed that respondents know the utility that they would receive from selecting one option of the choice set (Bateman et al., 2002). These two stated-preference methods (and their related statistical analysis) are frequently used to identify those demand–demand compromises associated with the different stakeholders' preferences and the socio-cultural factors underpinning them. Multi-criteria decision analysis would also help to integrate multiple stakeholders' perspectives (see Bryan et al., 2010).

Finally, Structural Equation Modeling (SEM; Grace, 2006) is a promising tool to investigate the causal relations between explanatory variables of change, ecosystem properties and the ES associations for supply or demand. SEM has been recently used to understand plant functional mechanisms underpinning ES supply compromises (Lavorel and Grigulis, 2012) and to evaluate the simultaneous effects of different explanatory variables of change on biodiversity, ES supply and human well-being (Santos-Martín et al., 2013).

As for the exploration of spatial ES associations, regression-based methods also provide an estimate of temporal associations. ARMA (AutoRegressive-Moving-Average) and derivate models make it possible to estimate the causality between the temporal trends of two quantitative ES. VAR (Vector AutoRegression) is the generalization of autoregressive models to more than one variable to explain each time series by its own lags and the lags of the other series. Further details on temporal autocorrelation are given in the appendix.

Beside classical regression models, other methods increasingly used for species distribution modeling such as machine-learning algorithms, should be preferred when the relationships among variables are complex, e.g. in the case of non-linearity responses or abrupt shifts (Leathwick et al., 2006). The most popular machine-learning methods are Random Forests (RF), Boosted Regression Trees (BRT), Artificial Neural Networks (ANN) and Bayesian Belief Networks (BBN). Among these, only BBN have been applied in the ES research because of their ability to incorporate uncertainty and to combine empirical data with expert knowledge (Landuyt et al., 2013), but few of them covered the analysis of ES associations (e.g. Ticehurst et al., 2007). Although BBNs offer the opportunity of analyzing the interactions between ES supply and demand, most studies do not include nodes with social or monetary values (Landuyt et al., 2013).

Most of these methods are not robust to collinearity (i.e. non-independence) among explanatory variables. Collinearity can introduce bias in the calculation of estimates and the ranking of predictors. It is particularly true when a model is built with data from one particular site or time step and transferred to another site or time step, for instance. Two main alternatives are available to limit collinearity, the use of "latent" variables (i.e. unobserved explanatory variables which encompass collinear ones) and the construction of aggregated variables from the collinear ones (see Dormann et al., 2013 for a complete review and methods to deal with collinearity). Once the biases are dealt with and models available, it possible to select the most parsimonious set of explanatory variables of ES bundles but more importantly to properly estimate the overall model performance. Optimization procedures are available for most methods listed above. It may be necessary to perform partial tests to discount for the effect of confounding factors, such as variables obviously driving ES indicators (e.g. climate) or variables structuring the modeling procedure (e.g. land cover classes). Finally, the relative influence of each potential determinant (e.g. environment, spatial component) can be estimated with a univariate or multivariate variance partitioning procedure (Borcard et al., 1992 and Gilbert and Bennett, 2010).

As this type of methods explores the explanatory variables of the ES bundles, they have been used for identifying either the ecological processes underpinning ES supply or the socio-cultural factors influencing ES demand, thus they appear as directly applicable to supply–supply, supply–demand and demand–demand cases (see Table 2).

4. Applications for ecosystem services assessment

Having reviewed the rich set of methods applicable to the identification and the understanding of supply–supply, supply–demand or demand–demand ES associations, in this final section we consider the challenges that these methods, and especially their combinations across disciplines, might help address.

Assessing current ES associations provides both a baseline against which to compare alternative future scenarios and insights into potential outcomes of policy and management decisions. Promoting multiple ES will entail reconciling ES trade-offs and enhancing synergies on both the supply- and demand-sides because socio-economic and ecological processes jointly drive ES bundles. In addition, feedbacks among supply and demand, like preferential management for the supply of ES with greater demand, necessitate the joint consideration of demand and supply of multiple ES, and their temporal dynamics, for policy design and land management (see Bryan, 2013 for example). As an example, such analyses may support regionally relevant choices between optimizing the supply of multiple ES at a given location (land sharing) or spatial segregation of ES supply (land sparing) because the spatial distribution of ES supply is subjected to socio-economical and/or ecological context (Willemen et al., 2012). One of the major challenges in the management of ES might thus be conciliating processes (i.e. ecological and socio-economic) occurring at diverse spatial and/or temporal scales (e.g. the Eurasian demand for soy products cause local trade-off between Amazonian forest conservation and soy production in Brazil). While the set of quantitative methods that we have reviewed have scarcely or never been applied, we contend that they offer an ideal toolbox to address such complexity and insure robust projections of ES supply and/or demand. Below, we briefly outline multiple sources of complexity that need to be incorporated into analyses.

First of all, the complex temporal and spatial ecological dynamics make it likely that relations among ES are not stationary in space and time. This is especially the case when (i) some ES are intensively managed until resource depletion (e.g. soil depletion in agricultural lands) or sensitive to climate change (e.g. decreasing tourism due to coral bleaching), (ii) when spatial trends in the supply of individual ES is context-dependent or (iii) when there are feedbacks between ecological functions (e.g. positive feedbacks among the production of easily decomposable plant material and soil fertility; Wardle et al., 2004). Second, feedbacks may also arise from management actions (e.g. a road network may directly alter habitat services through fragmentation and indirectly through a disrupted water supply and quality; Carpenter et al., 2009) or financial incentives supporting agro-ecosystems (Bryan and Crossman, 2013), and off-site effects from land use decisions at far away locations may alter local ES bundles (e.g. deviating water flows to maintain ecosystem functions and services may lead to water shortage and desiccation elsewhere; see Maestre Andrés et al., 2012 for other examples). Structural Equation Models, and by extension Path Analysis, appear to be one powerful way to integrate biophysical, management or demand feedbacks and yet, have only been used once to that end (Santos-Martín et al., 2013).

Third, as for spatial variations, temporal variations in ES supply and, even more in bundles, have also been scarcely studied. Depending on the mechanistic connections between services, the temporal variability in supply of a given ES supply may be determined by the variability of another ES and/or ES demand, making it essential to incorporate all types of ES associations into scenario modeling. While scenario analysis and modeling is frequently used for characterizing potential futures and assessing the consequences of different management options in ES associations, the temporal analysis of ES associations should go beyond scenario analysis through the inclusion of optimization algorithms (Seppelt et al., 2013). The optimization-based analysis can provide a set of optimum management solutions (i.e. Pareto frontier) in terms of ES associations in a social–ecological system (e.g. Lautenbach et al., 2013).

Fourth but not least, managing ES bundles needs to address how the ecological scale of ES supply matches the political and economic scales of decision-making. Only a few studies have explored the potential congruence or mismatch between the spatial scales of ES supply and ES management (e.g. Hein et al., 2006 and Willemen et al., 2012). To delineate the right ecological spatial scale, Luck et al. (2003) introduced the concept of service-providing units (SPUs), i.e. “ecosystem structures and processes that provide specific services at a particular spatial scale”, a concept that could be extended to any spatial unit supplying an ES bundle. Comparing such “bundle providing units” with scales of management might be the most relevant way to define at which scale trade-offs/synergies should be quantified and managed. Overall, scale-aware techniques (e.g. nested-downscaled modeling, network analysis, scenario analysis or time series methods) should be included in the methodological framework for analyzing ES associations (Scholes et al., 2013).

While the management of ES bundles is a priority for sustainability, the current focus on ES should not shadow the need to also protect biodiversity, as a baseline resource for ES as well as for its intrinsic value. The ethical issue of the prioritization of species or ecosystem processes essential for the supply of targeted ES in conservation and restoration planning has motivated analyses of the co-occurrence or complementarity among conservation strategies focusing on these different objectives (Chan et al., 2006, Bullock et al., 2011, Maes et al., 2012 and Cimon-Morin et al., 2013). Although relationships between biodiversity and ES are highly complex (Kremen, 2005, Balvanera et al., 2006 and Mace et al., 2012), understanding how changes in biodiversity and ecosystem properties alter ES supply remains a research priority (Nicholson et al., 2009, Cardinale, 2012 and Balvanera et al., 2014). The spatial congruence between biodiversity-rich areas and locations of ES supply has already been estimated by overlap analyses (e.g. Egoh et al., 2009). Several methods presented here, such as RDA, could give better insights into the relationships between biodiversity and ES.

5. Conclusion

ES delivery relies on complex interactions among ecological components, social components and landscape management, in which associations between ES can emerge not only on the supply-side (Rodriguez et al., 2006) but also on the demand-side (TEEB, 2010). Combining ecological (i.e. supply) and socio-economic (i.e. demand) aspects of ES relationships, three types of associations can be defined: the congruence between ES delivery ("supply–supply"), between ES supply and demand ("supply–demand") and among beneficiaries ("demand–demand"). Considering three main steps for analyses: (1) detecting ES associations, (2) defining ES bundles and (3) isolating explanatory variables behind ES associations, we have identified a broad spectrum of associated quantitative methods. While each method has its own strengths and weaknesses (Table 2) and results need to be interpreted in the light of these, we argue that assessing ES associations requires as much variety of techniques as complexity exists in specific case studies. Obviously, the choice of one method over another must be made carefully for consistency with the conceptual framework of the analysis, specific hypotheses to be tested and compatibility with data availability and scale. Therefore, for managing ES bundles in landscapes, where ES supply, ES demand and ES governance interact tightly, a diversity of methodological tools should be considered. These include not only those methods frequently used in the ES literature, such as star diagrams, overlapping maps of ES delivery, or correlation tests, but also alternative methods which have scarcely been applied (e.g. co-inertia analysis, GAM, decision trees and artificial neural networks, or distance approaches).

Managing landscapes for multiple ES raises the question of how ES trade-offs can be effectively mitigated and synergies enhanced. A methodological approach that considers a diverse range of methods to analyze ES associations, and uncovers the ecological and socio-economic factors driving ES bundles may be the only way to deal with the complexity of ES dynamics in socio-ecological systems.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:10.1016/j.gloenvcha.2014.07.012.

B. Synthesis of the consultative study on alpine iconic species

Enquête sur les espèces patrimoniales dans les Alpes Projet CONNECT

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1. Quelques éléments de contexte

Les Alpes françaises présentent un niveau de biodiversité généralement élevé, mais très variable selon les écosystèmes présents, le type d'usage des terres et les conditions biophysiques (climat – altitude …). Le but du projet CONNECT est d'examiner **comment différentes stratégies de « conservation de la nature » au sens large permettent de préserver la faune et la flore des Alpes, et plus généralement les différentes fonctionnalités écologiques des écosystèmes alpins**.

Traditionnellement, les politiques de conservation menées dans les Alpes ont été orientées « **biodiversité** », c'est-à-dire qu'elles ont pour but affiché de préserver une ou plusieurs espèce(s) donnée(s) (ou habitats spécifiques). Actuellement, un concept émergent pourrait infléchir ces politiques vers la conservation des zones à enjeu pour la fourniture de « **services écosystémiques** ». Entendus comme les bénéfices rendus par la nature aux hommes, ces services sont basés sur le fonctionnement écologique des milieux naturels et se traduisent par divers « rôles » identifiés par nos sociétés : limitation du risque d'érosion ou d'avalanche, maintien de la qualité de l'eau, production de bois, esthétique du paysage …

La question se pose aujourd'hui des **conséquences d'une gestion orientée « services écosystémiques » sur la biodiversité, et inversement**.

Toutefois, **définir « les zones à enjeu pour la biodiversité »** peut s'entendre de différentes manières, selon ce que l'on juge essentiel en termes de biodiversité, par exemple protéger un grand nombre d'espèces ou protéger des espèces particulières. C'est dans ce contexte que nous avons cherché à **établir une liste restreinte d'espèces** « qu'il semble particulièrement important » de préserver dans les Alpes. A ce titre, la consultation d'acteurs impliqués dans la gestion et la conservation de la nature alpine est une étape essentielle du processus.

A partir de ces définitions, trois zonages différents d'espaces à conserver en priorité se dessinent, correspondant à trois stratégies de conservation distinctes :

- Les espaces préservant **un maximum d'espèces**, quelles qu'elles soient,
- Les espaces préservant au mieux **un nombre restreint d'espèces particulières**,
- Les espaces préservant au mieux **la fourniture en services écosystémiques**.

Le but de notre étude est de déterminer les compatibilités et compromis entre des stratégies de conservation de la nature axées sur la conservation des espèces d'une part, et sur les services écosystémiques d'autre part.

2. L'enquête « Espèces patrimoniales dans les Alpes »

Notre but au travers de cette enquête est d'obtenir **une liste restreinte d'espèces** qui représentent des enjeux de conservation forts sur les Alpes. Ces espèces peuvent être choisies pour des **raisons diverses** (espèce parapluie – espèce à forte valeur culturelle – espèce menacée …).

A partir de l'ensemble des espèces animales vertébrées et des espèces végétales répertoriées en France, nous procédons en quatre étapes afin d'obtenir notre liste restreinte (voir Figure 1 ci-après) :

- Etape I : sélection des espèces **présentes** dans les Alpes,
- Etape II : sélection des espèces dont la répartition spatiale est **disponible** sur l'ensemble des Alpes,
- Etape III : sélection des espèces jugées **patrimoniales** pour les Alpes,
- Etape IV : espèces **sélectionnées** par les acteurs du territoire.

Notre sélection d'espèces se fait donc parmi **les espèces patrimoniales des Alpes**. De telles espèces, animales ou végétales, sont liées au territoire alpin dans la mesure où le maintien des populations dépend fortement de la conservation des milieux qu'elles occupent dans les Alpes. Les régions alpines portent donc une certaine « responsabilité » envers ces espèces patrimoniales, qu'on ne retrouve pas ou peu ailleurs en France. Sans bénéficier forcément d'un statut de protection officiel, ces espèces peuvent s'inscrire dans l'identité culturelle d'une région et sont le symbole de la biodiversité et du fonctionnement des écosystèmes tels que nous les connaissons, ou les avons connus.

Au niveau national, la Stratégie de Création de nouvelles Aires Protégées (SCAP) a été mise en œuvre suite aux Grenelles de l'Environnement. Elle vise à déterminer les espèces ciblées comme enjeu des aires protégées à venir, ce sont les espèces patrimoniales. Chaque région a ensuite repris cette liste pour établir au niveau régional quelles sont les espèces animales et végétales pour lesquelles le territoire porte une part importante de la responsabilité de leur conservation, et pour lesquelles un outil de protection surfacique est pertinent (aires protégées). Ce sont sur ces listes SCAP régionales que se base notre étude (étape de sélection III). Pour plus de détails sur la constitution de ces listes SCAP, consulter le site internet officiel http://scap.espaces-naturels.fr (Nom d'utilisateur : lecteur - Mot de passe : scapty).

Pour répondre à la question de l'étape IV, nous avons sollicité l'avis d'experts naturalistes, d'acteurs impliqués dans la conservation de la nature dans les régions PACA et Rhône-Alpes. Notre objectif a été d'obtenir une liste justifiée de 10 espèces animales vertébrées et 10 espèces végétales qui représentent des enjeux de conservation forts dans les Alpes.

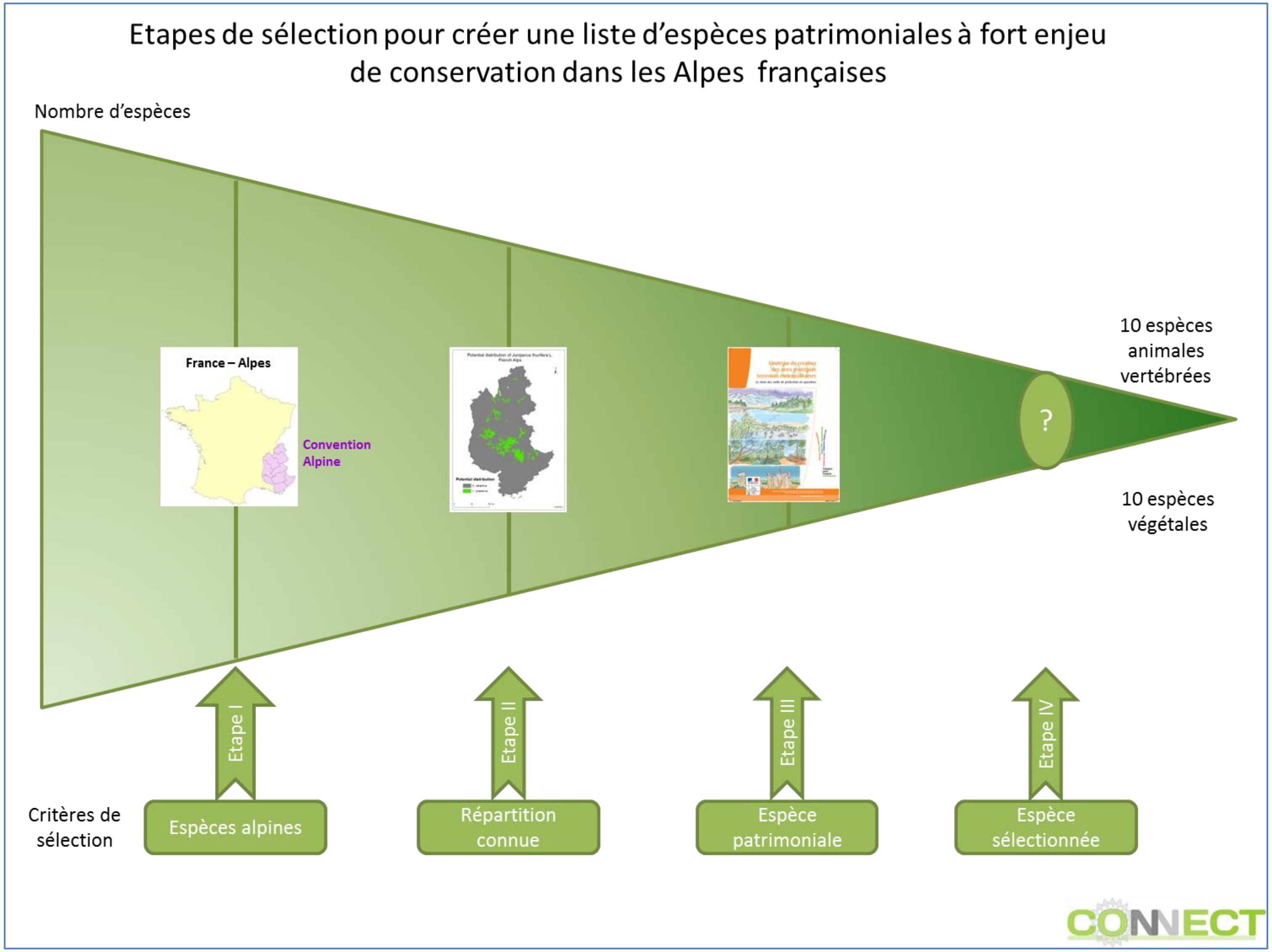

Figure 1 : Etapes et critères de sélection pour la création d'une liste restreinte d'espèces patrimoniales dans les Alpes françaises

3. Méthodologie générale

Distributions des espèces sur les Alpes

L'information de base nécessaire pour réaliser ce travail est d'avoir à disposition une carte représentant les zones de présence pour chacune des espèces sélectionnées.

Pour obtenir les distributions spatiales des espèces sur l'ensemble de notre zone d'étude, nous nous basons sur les données de **zone d'occurrence potentielle**. Cette zone est définie par l'IUCN (Union Internationale pour la Conservation de la Nature) comme « la superficie délimitée par la ligne imaginaire continue la plus courte possible pouvant renfermer tous les sites connus, déduits ou prévus de présence actuelle d'un taxon, à l'exclusion des individus erratiques. Cette mesure peut exclure des discontinuités ou disjonctions dans la répartition globale d'un taxon (par exemple de larges zones où l'habitat est, à l'évidence, inadéquat). » Il s'agit donc de **la zone géographique qu'une espèce donnée habite de manière habituelle et dans laquelle il est probable de la retrouver**.

Au terme du processus de modélisation mené par nos collaborateurs, nous disposons d'**une carte de distribution potentielle à la résolution de 1*1km par espèce** pour 380 vertébrés et 2 748 plantes vasculaires des Alpes. Chaque carte correspond à l'ensemble des milieux où les caractéristiques abiotiques sont favorables au maintien des populations.

Informations méthodologiques détaillées

*Une méthodologie détaillée expliquant la manière dont les aires de présence potentielle des espèces ont été déterminées est disponible sur simple demande. Les données pour les vertébrés sont issues des travaux de **Luigi Maiorano (DEE – University of Lausanne)** décrits dans l'article : [Maiorano L, Amori G, Capula M, Falcucci A, Masi M, et al. (2013) Threats from Climate Change to Terrestrial Vertebrate Hotspots in Europe. PLoS ONE 8(9): e74989. doi:10.1371/journal.pone.0074989]. Les données primaires pour les végétaux sont issues de deux inventaires provenant du **Conservatoire Botanique National Alpin** et du **Conservatoire Botanique National Méditerranée**. Ces données ont été retravaillées selon différents modèles de niches écologiques potentielles par les membres de **l'équipe EMABIO** du **LECA – CNRS**.*

Nous n'avons pas pu accéder à la distribution à l'échelle des Alpes d'espèces invertébrées (papillons, insectes…), malgré l'intérêt qu'elles représentent, c'est pourquoi elles sont absentes de notre étude.

Espèces patrimoniales – Données SCAP

Un des buts de notre étude est de cibler l'analyse sur un nombre d'espèces restreint, qui constituent donc des enjeux de protection pour les Alpes.

La loi n° 2009-967 du 3 août 2009 de programmation relative à la mise en œuvre du Grenelle de l'Environnement réaffirme la nécessité de protéger de nouveaux territoires terrestres et marins au travers de la Stratégie de Création de nouvelles Aires Protégées terrestres métropolitaines (SCAP). Pour aboutir à une couverture renforcée de la **richesse patrimoniale**, les travaux de la SCAP ont tenté de déterminer quels espèces, habitats et sites d'intérêt géologique on devait chercher à préserver en priorité dans le réseau français des aires protégées. Des indicateurs de richesse patrimoniale ont été recensés.

Une première liste comprend des espèces et habitats menacés ou pour lesquels la France a une responsabilité patrimoniale forte et pour lesquels un outil spatial de protection est pertinent.

Cette liste nationale a été élaborée en croisant plusieurs critères :

- Directive Habitats et Directive Oiseaux, et évaluation de l'état de conservation ;
- Listes rouges nationale et mondiale ;
- Espèces endémiques strictes ;
- Espèces et habitats déterminants ZNIEFF ;
- Espèces bénéficiant ou ayant bénéficié d'un plan national d'actions ;
- Quelques espèces potentiellement sensibles aux changements climatiques ;
- 30 espèces d'invertébrés souterrains.

Cette liste a ensuite été déclinée de manière régionale, de façon à cibler plus spécifiquement les territoires capables de maintenir au mieux les populations d'espèces concernées.

La SCAP répond aux questions suivantes :

- **Quelles espèces et quels types d'habitats doit-on chercher à préserver en priorité par un réseau d'aires protégées ?**
  ➔ Telle est la finalité de la construction des listes « espèces et habitats » menacés pour lesquels la responsabilité patrimoniale de la France est forte et pour lesquels un outil de protection surfacique est pertinent.
- **Quelles sont, parmi ces espèces et ces habitats, ceux pour lesquels le réseau d'espaces protégés existants n'est pas suffisant ?**
  ➔ Tel est l'intérêt du diagnostic patrimonial du réseau des aires protégées qui a abouti à la constitution de fiches par espèces décrivant la répartition de l'espèce ainsi que son statut national.
- **Pour un habitat ou une espèce non encore suffisamment protégé, quels sont les espaces qu'il faudra protéger et suivant quelles modalités particulières ?**
  ➔ Tel est l'objectif des déclinaisons régionales qui listent les espèces pour lesquelles la responsabilité patrimoniale de la région est forte et pour lesquelles un outil de protection surfacique est pertinent.

Les listes régionales SCAP contiennent donc un ensemble d'espèces « qu'il semble important de protéger en priorité » dans les Alpes, sur la base de critères explicites, validés par différents experts et reconnus au niveau national. Nous avons considéré les espèces présentes dans les listes régionales de Rhône-Alpes et Provence-Alpes-Côte d'Azur pour la suite de notre étude.

Bilan des effectifs avant sélection par les acteurs

Nombre d'espèces concernées		Espèces alpines patrimoniales (SCAP)	Espèces alpines à répartition disponible
Faune	Amphibiens	5	**5**

Faune	Reptiles	7	**6**
Faune	Mammifères	13	**13**
Faune	Oiseaux nicheurs	72	**40**
Faune (vertébrés)			**64**
Flore	Dicotylédones	35	**20**
Flore	Monocotylédones	15	**5**
Flore	Gymnospermes	1	**1**
Flore	Ptéridophytes	12	**5**
Flore (plantes vasculaires)			**31**

Tableau 1 : Bilan des effectifs des espèces conservées par les étapes de sélection pour l'enquête "Espèces patrimoniales dans les Alpes"

Sélection par les acteurs du territoire

Vingt-et-une structures différentes ont été sollicitées début 2013 sur Rhône-Alpes et PACA parmi les acteurs impliqués dans la conservation de la nature et les experts naturalistes (parcs naturels – conservatoires d'espaces naturels – laboratoires scientifiques de recherche en biologie – associations naturalistes…). Notre demande portait sur la sélection justifiée d'un nombre très restreint d'espèces à enjeu fort de conservation, en vue de rendre compte des conséquences pour ces espèces particulières et pour la richesse de plantes et de vertébrés de différentes stratégies de conservation.

Au terme du processus d'enquête, douze réponses favorables ont été rendues, plus particulièrement en provenance du monde de la recherche et des structures officielles de conservation de la nature. Deux de ces réponses n'ont pas pu être exploitées car elles concernaient des espèces invertébrées uniquement, pour lesquelles nous ne disposons pas des distributions spatiales à l'échelle des Alpes.

Sur la base d'entretiens en direct ou d'échanges écrits, et parmi la liste d'espèces proposée, nous avons demandé aux experts enquêtés de sélectionner 10 vertébrés et 10 plantes de manière « prioritaire », en ce sens que leur conservation sur les Alpes leur paraît particulièrement intéressante.

Des raisons très diverses ont été évoquées pour justifier de l'intérêt particulier de chacune des espèces sélectionnées. Nous avons classé les arguments proposés pour la sélection de chacune des espèces selon la liste suivante :

- Espèce bénéficiant d'un **statut de protection** particulier
- Valeur **affective** forte
- Valeur **utilitaire** (cueillette par exemple)
- Valeur **esthétique**
- **Rôle fonctionnel** clé dans l'écosystème
- **Effectif faible** à soutenir

- Espèce **parapluie**
- Espèce **endémique**
- Espèce **emblématique**
- **Autre**

4. Résultats

Espèces et critères de sélection

Le tableau 2 ci-dessous liste les 21 espèces qui ont reçu le plus de votes : 10 espèces animales vertébrées et 11 espèces végétales (poussées à 11 par égalité de scores).

Vertébrés		**Plantes vasculaires**	
Nom latin	Nom courant	*Nom latin*	Nom courant
Vipera ursinii	Vipère d'Orsini	*Eryngium alpinum L.*	Panicaut des Alpes, Etoile des Alpes
Lynx lynx	Lynx boréal	*Astragalus alopecurus Pall.*	Queue de renard des Alpes
Lutra lutra	Loutre d'Europe	*Dracocephalum austriacum L.*	Dracocéphale d'Autriche
Rhinolophus hipposideros	Petit rhinolophe	*Cypripedium calceolus L.*	Sabot de Vénus
Speleomantes strinatii	Spélerpès de Strinati	*Juniperus thurifera L.*	Genévrier thurifère
Lepus timidus	Lièvre variable	*Liparis loeselii (L.) Rich.*	Liparis de Loesel
Gypaetus barbatus	Gypaète barbu	*Aquilegia alpina L.*	Ancolie des Alpes
Hieraaetus fasciatus	Aigle de Bonelli	*Potentilla delphinensis Gren. & Godr.*	Potentille du Dauphiné
Tetrao tetrix	Tétras lyre	*Saxifraga florulenta Moretti*	Saxifrage à nombreuses fleurs
Aegolius funereus	Nyctale de Tengmalm, Chouette de Tengmalm	*Serratula lycopifolia (Vill.) A.Kern.*	Serratule à feuilles de chanvre d'eau
		Marsilea quadrifolia L.	Fougère d'eau à quatre feuilles, Marsilea à quatre feuilles

Tableau 2 : Espèces sélectionnées dans le cadre de l'enquête "Espèces patrimoniales prioritaires dans les Alpes"

Pour la sélection des espèces animales, l'argument le plus souvent cité est lié à des **effectifs faibles** de population dans les Alpes, qu'il s'agirait ainsi de soutenir. Ensuite, un critère de sélection très souvent avancé est celui des **espèces parapluie**, dont la conservation entraînerait de fait celle d'un important cortège d'espèces liées, ou dont la présence témoigne de la qualité de l'écosystème. Les arguments fonctionnels sont également sollicités, puisque le **rôle fonctionnel** important de certaines espèces est mis en avant (prédateur – charognard …).

En ce qui concerne la sélection parmi les espèces végétales, les arguments utilisés diffèrent. Le **caractère emblématique** des espèces revient comme premier marqueur de sélection, suivi par la **valeur esthétique** accordée aux différentes plantes. Le **statut de protection** actuel des espèces soutient également le choix de certaines espèces.

Il est intéressant de noter la **différence entre arguments proposés** entre la sélection d'espèces animales et végétales. Les vertébrés sont discriminés selon des critères scientifiques essentiellement (effectifs et aspects fonctionnels) alors que les arguments subjectifs l'emportent lorsque ce sont des espèces végétales qui sont étudiées (valeurs emblématique et esthétique).

Valorisation et limites de l'étude

Le but de construction de cette liste restreinte d'espèces patrimoniales particulières est d'inclure dans les stratégies de conservation étudiées une **dimension culturelle** aux données de biodiversité. En effet, le choix de certaines espèces parmi celles dont la distribution est disponible est fonction de différents critères, à la fois objectifs et subjectifs.

Force est de constater le **faible nombre de répondants** ayant participé à cette étude. Dix réponses ont pu être exploitées uniquement, elles émanent essentiellement du domaine de la recherche en biologie ou de structures officielles de conservation de la nature (conservatoire botanique – conservatoire d'espaces naturels). Notre sollicitation a trouvé peu d'écho auprès des associations de protection de la nature. Ce faible taux de réponse semble lié à deux facteurs. Le premier serait la non-anticipation de notre part du **poids politique** donné à cette sélection. Ainsi, se concentrer sur 20 espèces pour proposer un scénario de conservation a semblé largement insuffisant, voire dangereux. Nous n'avions en effet pas anticipé la crainte liée à une récupération politique d'une telle liste d'espèces, mais souhaitions simplement illustrer les scénarios de conservation de la nature par un cas extrême de sélection. Le second facteur limitant semble lié au fait de proposer une sélection uniquement sur les espèces des **listes SCAP**, car leur pertinence a été questionnée à plusieurs reprises.

En conséquence, nous n'avons pas souhaité réutiliser directement les résultats de l'enquête présentée dans l'analyse de différentes stratégies de conservation. La réflexion sur les espèces patrimoniales des Alpes pourra être reprise ultérieurement, mais dans le cadre de notre étude, nous avons choisi de baser la prise en compte de la valeur culturelle de la biodiversité sur une liste d'espèces officielle et déjà constituée. Ainsi, nous avons considéré l'ensemble des espèces classées par la **liste rouge nationale** de l'UICN (Union Internationale pour la Conservation de la Nature) selon les catégories « En danger critique d'extinction (CR) », « Espèce en danger (EN) » et « Espèce vulnérable (VU) ». La liste finale des espèces dont la distribution est disponible contient 45 plantes, 7 reptiles, 7 amphibiens, 10 mammifères et 83 oiseaux (liste jointe Section V. Documents annexes).

5. Documents annexes

Liste des vertébrés et plantes classées par la liste rouge nationale de l'UICN selon les catégories « En danger critique d'extinction (CR) », « Espèce en danger (EN) » et « Espèce vulnérable (VU) ». Seules les espèces dont la distribution spatiale était disponible ont été conservées.

Catégorie	Nom scientifique	Statut Liste Rouge Nationale
Plante	Achillea moschata Wulfen	VU
Plante	Adonis pyrenaica DC.	VU
Plante	Aethionema thomasianum Gay	VU
Plante	Androsace septentrionalis L.	VU
Plante	Artemisia atrata Lam.	VU
Plante	Astragalus alopecurus Pallas	EN
Plante	Astragalus leontinus Wulfen	VU
Plante	Bifora testiculata (L.) Sprengel in Schultes	EN
Plante	Carduus aurosicus Chaix	VU
Plante	Carex atrofusca Schkuhr	VU
Plante	Carex firma Host	VU
Plante	Carex melanostachya M. Bieb. ex Willd.	VU
Plante	Carex microglochin Wahlenb.	VU
Plante	Chamorchis alpina (L.) L.C.M. Richard	VU
Plante	Cortusa matthiolii L.	VU
Plante	Cotoneaster delphinensis Chatenier	VU
Plante	Crepis rhaetica Hegetschw.	VU
Plante	Cypripedium calceolus L.	VU
Plante	Cytisus ardoini E. Fourn.	VU
Plante	Dactylorhiza incarnata (L.) So¢	VU
Plante	Danthonia alpina Vest	EN
Plante	Doronicum clusii (All.) Tausch subsp. clusii	VU
Plante	Draba hoppeana Reichenb. in Moessler	VU
Plante	Dracocephalum austriacum L.	VU
Plante	Euphorbia peplus L.	VU
Plante	Genista delphinensis Verlot (b.)	VU
Plante	Gentianella ramosa (Hegetschw.) Holub	VU
Plante	Geranium argenteum L.	VU
Plante	Gymnadenia odoratissima (L.) L.C.M. Richard	VU
Plante	Hierochloë odorata (L.) P. Beauv. subsp. odorata	VU
Plante	Leucanthemum burnatii Briq. & Cavillier	VU
Plante	Liparis loeselii (L.) L.C.M. Richard	VU
Plante	Potentilla delphinensis Gren. & Godron	VU
Plante	Saussurea discolor (Willd.) DC.	VU
Plante	Saxifraga florulenta Moretti	VU
Plante	Saxifraga valdensis DC.	VU
Plante	Serratula lycopifolia (Vill.) A. Kerner	VU

Plante	Sisymbrium strictissimum L.	VU
Plante	Smyrnium perfoliatum L.	VU
Plante	Spiranthes aestivalis (Poiret) L.C.M. Richard	VU
Plante	Tofieldia pusilla (Michaux) Pers. subsp. pusilla	VU
Plante	Trifolium saxatile All.	VU
Plante	Tulipa raddii Reboul	EN
Plante	Valeriana celtica L.	VU
Plante	Viola pinnata L.	VU

Catégorie	Nom scientifique	Statut Liste Rouge Nationale
Mammifère	Lynx lynx	EN
Mammifère	Miniopterus schreibersi	VU
Mammifère	Mustela lutreola	EN
Mammifère	Myotis capaccinii	VU
Mammifère	Myotis punicus	VU
Mammifère	Ovis orientalis	VU
Mammifère	Rhinolophus mehelyi	CR
Mammifère	Ursus arctos	CR
Mammifère	Canis lupus	VU
Mammifère	Cricetus cricetus	EN
Amphibien	Pelobates cultripes	VU
Amphibien	Pelobates fuscus	EN
Amphibien	Rana arvalis	CR
Amphibien	Rana pyrenaica	EN
Amphibien	Salamandra atra	VU
Amphibien	Bombina variegata	VU
Amphibien	Salamandra lanzai	CR
Reptile	Iberolacerta aranica	EN
Reptile	Iberolacerta aurelioi	CR
Reptile	Iberolacerta bonnali	EN
Reptile	Timon lepidus	VU
Reptile	Mauremys leprosa	EN
Reptile	Testudo hermanni	VU
Reptile	Vipera ursinii	CR

Catégorie	Nom scientifique	Statut Liste Rouge Nationale
Oiseau	Acrocephalus arundinaceus	VU
Oiseau	Acrocephalus paludicola	VU
Oiseau	Aegypius monachus	CR
Oiseau	Alca torda	CR

Oiseau	Anas crecca	VU
Oiseau	Anas querquedula	VU
Oiseau	Anser anser	VU
Oiseau	Anser fabalis	VU
Oiseau	Anthus pratensis	VU
Oiseau	Aquila chrysaetos	VU
Oiseau	Asio flammeus	VU
Oiseau	Bonasa bonasia	VU
Oiseau	Botaurus stellaris	VU
Oiseau	Calonectris diomedea	VU
Oiseau	Carduelis cannabina	VU
Oiseau	Charadrius hiaticula	VU
Oiseau	Chlidonias niger	VU
Oiseau	Ciconia nigra	EN
Oiseau	Circus aeruginosus	VU
Oiseau	Circus pygargus	VU
Oiseau	Columba livia	EN
Oiseau	Crex crex	EN
Oiseau	Cygnus columbianus	EN
Oiseau	Dendrocopos leucotos	VU
Oiseau	Elanus caeruleus	EN
Oiseau	Emberiza hortulana	EN
Oiseau	Falco naumanni	VU
Oiseau	Fratercula arctica	CR
Oiseau	Galerida theklae	VU
Oiseau	Gallinago gallinago	EN
Oiseau	Gavia immer	VU
Oiseau	Gelochelidon nilotica	VU
Oiseau	Glareola pratincola	EN
Oiseau	Glaucidium passerinum	VU
Oiseau	Grus grus	CR
Oiseau	Gypaetus barbatus	EN
Oiseau	Hieraaetus fasciatus	EN
Oiseau	Hieraaetus pennatus	VU
Oiseau	Hippolais icterina	VU
Oiseau	Hirundo daurica	VU
Oiseau	Lanius excubitor	EN
Oiseau	Lanius meridionalis	VU
Oiseau	Lanius minor	CR
Oiseau	Larus audouinii	EN
Oiseau	Larus canus	VU
Oiseau	Larus genei	EN
Oiseau	Limosa limosa	VU
Oiseau	Locustella luscinioides	EN
Oiseau	Melanitta fusca	EN

Oiseau	Melanocorypha calandra	EN
Oiseau	Mergus albellus	VU
Oiseau	Milvus milvus	VU
Oiseau	Muscicapa striata	VU
Oiseau	Neophron percnopterus	EN
Oiseau	Numenius arquata	VU
Oiseau	Numenius phaeopus	VU
Oiseau	Oenanthe hispanica	EN
Oiseau	Pandion haliaetus	VU
Oiseau	Phoenicopterus roseus (ruber)	EN
Oiseau	Phylloscopus sibilatrix	VU
Oiseau	Picus canus	VU
Oiseau	Platalea leucorodia	VU
Oiseau	Podiceps auritus	VU
Oiseau	Porphyrio porphyrio	EN
Oiseau	Porzana parva	CR
Oiseau	Porzana pusilla	CR
Oiseau	Pterocles alchata	CR
Oiseau	Puffinus mauretanicus	VU
Oiseau	Puffinus puffinus	VU
Oiseau	Puffinus yelkouan	VU
Oiseau	Pyrrhula pyrrhula	VU
Oiseau	Remiz pendulinus	EN
Oiseau	Saxicola rubetra	VU
Oiseau	Sitta whiteheadi	VU
Oiseau	Somateria mollissima	CR
Oiseau	Stercorarius longicaudus	VU
Oiseau	Sterna dougallii	CR
Oiseau	Sterna paradisaea	CR
Oiseau	Sterna sandvicensis	VU
Oiseau	Sylvia conspicillata	EN
Oiseau	Tetrao urogallus	VU
Oiseau	Tetrax tetrax	VU
Oiseau	Uria aalge	EN

C. Supporting Information S1

1. S1.A Ecological parameters complementary description

In respond to the call for more formalised description of variables used in ecological assessments (Martínez-Harms, M.J. & Balvanera, P. 2011; Crossman et al. 2013), we proposed a short description and additional information on the 18 ecological parameters modelled and analysed (Table S1.1).

1) Agricultural production:
    - Aggregation of actual yields for annual crops, vineyards and orchards for 2009, from official statistics at the "département" level.
    - Initial range: [0 – 33 222] kg/ha/year

2) Forage production:
    - Aggregation of yields of pastures, meadows and mountain grasslands, defined at the level of the "département" for 2009. Yields for each kind of pasture, meadow or mountain grassland were refined according to their likely presence depending on altitude in a given eco-region.
    - Initial range: [0 – 4998] kg of dry matter/ha/year

3) Wood production:
    - Potential woody biomass supply estimated for 2010 for stemwood and logging residues. Theoretical biomass potential was estimated from recent, detailed forest inventory data using the EFISCEN model and corresponds to bio-physical potentials of the forests. Constraints reducing the availability of woody biomass were defined and quantified regarding social, technical and environmental aspects to assess the realizable potential. Data were disaggregated from statistical regions to grid level based on spatially-explicit data on tree species.
    - Initial range: [0 – 1.26] Gg dry matter/year/km^2

4) Hydro-energy potential:
    - Potential hydro-energy power delivered by river basin, using five classes. This index reflects the potential amount of energy that could be produced according to physical assets of the region (e.g. slope – rivers length and flow). Biophysical characteristics of the basin impact hydro-energy potential by modulating rainfall and the runoff volumes, as well as vegetation uptake.
    - Initial range: [0 – 227 000] kW

5) Recreation potential:
    - Combination of three components to represent what ecosystems potential offer for daily recreation: degree of naturalness; protected areas; distance from coast and water quality. On the "potential flow" firstly accessibility was estimated, then a Recreation Opportunity Spectrum (ROS) was built thanks to expert contribution to define accessibility, and areas of different provision/accessibility were obtained. Finally statistics were derived on which amount of population has access to which type of ROS zones.
    - Initial range: [0 – 0.89] adimensional index

6) Territorial capital for rural tourism:
    - Potential for ‘rural tourism’ incorporating the supply of ‘beach tourism’, of attractions for winter tourism, of attractions for nature tourism and assets of symbolic capital. The capacity for rural tourism is defined as the ability of the region to provide tourist activities that take place outside urban areas and involve overnight stays. The concept of territorial capital is employed to integrate environmental and human capacities when assessing rural development potentials.
    - Initial range: [0 – 0.74] adimensional index

7) Leisure hunting:
    - Number of wild ungulates killed in one hunting period, by species and zones, converted into the number of killed animal per km² of each zone and adding results for all species. By using actual hunting bags this definition includes the ability of ecosystems to host biodiversity, and societal demand game.
    - Initial range: [0 – 21] number of animals/km²/year

8) Protected plant species:
    - Overlay of potential ecological niche distributions for the 45 protected plants species hosted by the French Alps for which potential distributions were available. Protected species are those with IUCN French Red List status critical, endangered and vulnerable. (see Biodiversity parameters)
    - Initial range: [0 – 11] number of species/km²

9) Protected vertebrate species:
    - Overlay of potential ecological niche distributions for the 107 protected vertebrate species hosted by the French Alps for which potential distributions were available. Protected species are those with IUCN French Red List status critical, endangered and vulnerable. (see Biodiversity parameters)
    - Initial range: [0 – 26] number of species/km²

10) Erosion mitigation:
    - Ability of biotic factors to decrease erosion risk i.e. difference between potential risk class (ignoring vegetation role) and effective risk class (including vegetation role). Potential and effective risks were determined using the empirical model RUSLE adapted to the Alps conditions.
    - Initial range: [1 – 5] adimensional index

11) Protection against rockfalls:
    - Ability of forests to decrease rockfall hazard i.e. presence of forests susceptible of intercepting or slowing rocky projectiles between probable starting points and actual sensitive areas linked to human infrastructures and presence. Specific forestry model RockForLIN and computer utility RollFree were used.
    - Initial range: [0 – 1 716] adimensional index

12) Chemical water quality regulation:
    - Amount of nitrogen retained by river basin. The model considers the input of diffuse and point sources of total nitrogen and estimates the nitrogen fraction retained during the transport from land to surface water (basin retention) and the nitrogen fraction retained in the river segment (river retention). The statistical proxy modelling uses GREEN model. In order to get a surface index

showing the contribution of the whole river basin, all linear indexes were averaged by river basin as final index.
- Initial range: [0 – 120] tN/km/year

13) Physical water quantity regulation:
    - Landscape capacity to modify the river discharge after heavy precipitation events potentially causing flood events, compared to a "worst case" scenario in terms of water retention regarding soil and land uses potential combinations. This index is based on the variability of the peak discharge at the outlet of a catchment in dependence of land use and soil distribution. The proxy modelling uses the hydrological model STREAM.
    - Initial range [0 – 1] adimensional index

14) Biological control of pests:
    - Overlay of potential ecological niche distributions for 110 vertebrate species providing natural control of invertebrate and rodent pests. (see Biodiversity parameters)
    - Initial range: [0 – 63] number of species/km²

15) Pollination:
    - Relative capacity of ecosystems to support crop pollination, in relation to the availability of floral resources, bee flight ranges and the availability of nesting sites.
    - Initial range: [0 – 0.7] adimensional index

16) Carbon storage:
    - Aggregation of carbon stocks from above-ground biomass, below-ground biomass, dead organic matter and soils, using the InVEST platform, module Carbon.
    - Initial range: [0 – 284] tC/ha

17) Plant diversity:
    - Overlay of potential ecological niche distributions refined with presence data and habitat preferences to better fit actual distributions for the 2748 plants species hosted by the French Alps for which potential distributions were available. Primary field data were used to model ecological niche distributions based on biophysical information.
    - Initial range: [0 – 776] number of species/km²

18) Vertebrate diversity:
    - Overlay of potential ecological niche distributions refined with presence data and habitat preferences to better fit actual distributions for the 380 vertebrate species hosted by the French Alps for which potential distributions were available. For each species, spatially explicit information on the extent of occurrence was collected from various sources. A suitability score was assigned by experts and literature to land cover classes to distinguish land-use/land-cover classes that represent suitable from inadequate habitats. Elevation range where each species can be found and maximum distance to water were combined with habitat suitability scores to refine the available extents of occurrence, as well as all freely available presence points.
    - Initial range: [0 – 227] number of species/km²

Table S1.1: Formalized description of ecological parameters modelled and analysed

Parameter	**Modelling method**		**Type of data**			**Parameter characteristics**				**Sources**
	Initial extent (initial resolution)	**Type of method**	**Bio physical**	**Socio-economic**	**Mixed**	**Actual / Potential**	**Beneficiary** (supply / demand)	**Type** (stock / flow / status)	**Direct / Proxy**	
Agricultural production (crop)	Département (100*100m)	Extrapolation of primary data	X		X	Actual	Supply * Demand	Flow	Direct	Agreste 2009
Forage production (fodd)	Département (100*100m)	Extrapolation of primary data + Lookup tables	X		X	Actual	Supply * Demand	Flow	Direct	Agreste 2009 - Supporting Information S1.B
Wood production (wood)	Europe (1*1km)	Causal relationships	X	X	X	Potential	Supply * Demand	Flow	Direct	Verkerk *et al.* 2011; Brus *et al.* 2012; Elbersen *et al.* 2012
Hydro-energy potential (hydro)	Rhône Méditerrannée watershed (river basin: mean area= 135 km²)	Extrapolation of primary data	X			Potential	Supply	Flow	Direct	Agence de l'eau RMC 2008
Recreation potential (recre)	Europe (1*1km)	Causal relationships + Expert knowledge	X	X	X	Potential	Supply * Demand	Status	Proxy	Paracchini *et al.* 2014
Tourism (tour)	Europe (1*1km)	Causal relationships + Expert knowledge	X	X	X	Potential	Supply * Demand	Status	Proxy	Paracchini & Capitani 2011; Maes *et al.* 2012, Paracchini *et al.* 2014
Leisure hunting (hunt)	Département (downscaled to 1*1km)	Extrapolation of primary data		X	X	Actual	Supply * Demand	Flow	Direct	Convention with « Réseau Ongulés Sauvages ONCFS / FNC / FDC » Supporting Information S1.C
Protected plant species (protp)	French Alps (250*250m)	Causal relationships + Extrapolation of primary data	X		X	Potential	Supply * Demand	Stock	Direct	Thuiller *et al.* 2014
Protected vertebrate species (protv)	Europe (1*1km)	Expert knowledge + Extrapolation of primary data + Lookup tables	X		X	Potential	Supply * Demand	Stock	Direct	Maiorano et al. 2013
Erosion mitigation (eros)	Alps (100*100m)	Causal relationships	X		X	Actual	Supply	Status	Proxy	Bosco *et al.* 2008; Bosco *et al.* 2009
Protection against rockfalls (rock)	French Alps (50*50m)	Causal relationships	X	X	X	Actual	Supply * Demand	Status	Proxy	Berger *et al.* 2013

Chemical water quality regulation (wql)	Europe (1*1km)	Causal relationships	X	X	X	Actual	Supply	Flow	Proxy	Grizzetti & Bouraoui 2006
Physical water quantity regulation (wqt)	Europe (1*1km)	Causal relationships + Lookup tables	X		X	Actual	Supply	Status	Proxy	Stürck, Poortinga & Verburg 2014
Biological control of pests (cbiol)	Europe (1*1km)	Expert knowledge + Extrapolation of primary data + Lookup tables	X		X	Potential	Supply	Stock	Proxy	Civantos et al. 2012; Maiorano et al. 2013
Pollination (poll)	Europe (1*1km)	Causal relationships + Lookup tables	X		X	Potential	Supply	Status	Proxy	Zulian, Maes & Paracchini 2013
Carbon storage (csto)	French Alps (100*100m)	Causal relationships + Lookup tables	X		X	Actual	Supply	Stock	Direct	Martin et al. 2011; Meersmans et al. 2012a, 2012b; Supporting Information S1.D
Plant diversity (plant)	French Alps (250*250m)	Causal relationships + Extrapolation of primary data	X		X	Potential	Supply	Stock	Direct	Thuiller *et al.* 2014
Vertebrate diversity (vert)	Europe (1*1km)	Expert knowledge + Extrapolation of primary data + Lookup tables	X		X	Potential	Supply	Stock	Direct	Maiorano *et al.* 2013

The **type of methods** was characterized according definitions of Martínez-Harms & Balvanera 2012, which are the following:

- Lookup tables: Use of existing ES values from the literature to land cover classes
- Expert Knowledge: Experts rank land cover types based on their potential to provide specific ES
- Causal relationships: Incorporate existing knowledge about how different layers of information related to ecosystem processes and the services to create a new proxy layer of the ES
- Extrapolation of primary data: Field data databases weighted by cartographical data (generally land cover)
- Regression models: Employing field data of ecosystem services as response variables and proxies (e.g. biophysical data and other sources of information obtained from GIS) as explanatory variables.

The **type of data** was characterized according definitions of Martínez-Harms & Balvanera 2012, which are the following:

- Biophysical data: Land cover, remote sense, topographical, hydrological and climate data
- Socio-economic data: Road map, population map, photos and census data
- Mixed data: databases, field data, surveys and bibliography

Parameter characteristics were partly characterized so as to fill in the blueprint proposed by Crossman *et al*. 2013, and were defined as follow:

- Actual / Potential: an actual parameter represents the functioning of the ecosystems and the way human benefit from it in reality (e.g. agricultural productions are an actual service as we used real statistics on volume harvested). A potential parameter represents the functioning of ecosystems and the way human could benefit from it, regardless of real uses (e.g. hydro-energy potential is a potential service as we used data on water flow power, regardless of the existence of hydro-energy plants in reality).
- Beneficiary: depending on the side of the ecosystem service cascade informed (Haines-Young & Potschin 2010), parameters can relate to the supply of the ecosystem service (biophysical side) or to the demand side (socio-cultural side), or to a combination of both sides (considering biophysical and socio-cultural attributes).
- Parameter type: the parameter can represent a stock (e.g. number of species/km^2), a flow (e.g. tons of wood harvested/year) or a status (e.g. relative capacity to buffer floods or to host pollinators).
- Direct / Proxy: a direct variable informs fully about the parameter (e.g. the total number of protected species hosted by an ecosystem directly relates to the cultural service protected species richness). A proxy variable informs partially about the parameter and is usually chosen as the direct variable is unknown or too difficult to access to (e.g. nitrogen retention capacity acts as a proxy for water quality regulation, as the actual characterization of the parameter would require additional inputs, linked for instance to pollutants retention, which would be complex to integrate all).

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2. S1.B Forage production: methodological information

Different grassland types have been considered based on their varying productive capacities. Thus, we distinguished between artificial grasslands, temporary grasslands, permanent grasslands and grasslands of very low productivity. We mapped them according Corine Land Cover 2006 categories refined by data on the probability of finding grassland types by altitude and eco-regions as described in local vegetation guides.

For grasslands up to 1500 m, we used yield data coming from agricultural statistics per département (Agreste 2009), weighted by the proportional area of each grassland type per altitude in each department (Equation 1).

For grasslands above 1500 m, we used yield data from five vegetation guides describing the main features of grasslands in the Alps. We averaged yields of typical grasslands per zone and altitude to provide a synthetic value of common yields (Equation 2).

Equation 1 (up to 1500m)

$$Mean\ yield_{[i,j]} = \frac{1}{S_{Tot\ [i,j]}}\left(Y_{AG} * \%S_{AG[i,j]} * S_{AG} + Y_{TG} * \%S_{TG[i,j]} * S_{TG} + Y_{PG} * \%S_{PG[i,j]} * S_{PG} + Y_{LPG} * \%S_{LPG[i,j]} * S_{LPG}\right)$$

Equation 2 (above 1500 m)

$$Mean\ yield_{[i',j']} = Average\ yield\ of\ possibly\ existing\ grassland\ types_{[i',j']}$$

With:
- i = altitudinal range up to 1500m: [0-1000m], [1000-1500m]
- i'= altitudinal range above 1500m: [1500-3000m per steps of 100m]
- j = département
- j' = eco-region
- Y = yield of each kind of grassland per département (tDM/ha), from Agreste
- *Grassland type* = artificial AG, temporary TG, permanent PG and of very low productivity LPG
- S = surface area (ha) of each grassland type per département, from Agreste
- S_{Tot} = surface of all grassland types per département, from Agreste
- $\%S[i,j]$ = percentage of a type of grassland for an altitudinal range and a zone.

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3. S1.C Leisure hunting: methodological information

Primary data was courtesy of the French National Office of Hunting and Wild animals (ONCFS) and of the National and Departmental Federations of Hunters (FNC / FDC).

They consisted in the number of animal actually killed during one hunting period in a given area (Table S1.2).

Table S1.2: Hunting data characteristics per species

Game species	Aggregation scale	Year
Red deer	Hunting zones	2010
Chamois	Département	2010
Corsican and Mediterranean mouflon	Département	2010
Roe deer	Municipality	2008
Wild boar	Municipality	2012

In order to build a unique indicator, the number of animals killed per species and aggregation scale (hunting zones, “département” or municipality) was converted into a number of killed animals per km² by dividing by each zone surface. Then, we summed all ungulates killed per km², whatever species each individual was belonging to. As such, all species are given an equal weight; we do not consider possible hunters’ preferences for one or the other species: only the overall number of ungulates killed is shown.

Datasets from different years were aggregated as comparison of available statistics confirmed the overall stability of hunting trends over recent years.

4. S1.D Carbon storage: methodological information

The ES represents the ecosystem actual stock of organic in four compartments: above-ground biomass (AGB), below-ground biomass (BGB), dead organic matter and soils. We investigated only more significant compartments by land cover types (Table S1.3).

Table S1.3: Compartments investigated for organic carbon stock assessment

	AGB	BGB	Dead organic matter (litter)	Soils
Arable lands	**No** (harvested and consumed almost every year)	**No** (considered not significant)	**No** (considered not significant)	**Yes** (directly extracted from the work presented in Meersmans et al. 2012 a,b and aggregated at 1*1 km)
Grasslands and open spaces	**No** (harvested and consumed almost every year)	**Yes** (estimated from AGB via conversion with root-shoot ratio of 1)	**No** (considered not significant – lack of relevant data)	
Forests	**Yes** (from harvestable volumes to total volumes via conversion factors)		**Yes** (national estimates)	

D.1 Grasslands

<u>Step 1: From AGB to BGB</u>

Grasslands BGB is not directly available, as it is never harvested or used by people. We used the root-shoot ratio to convert AGB data to BGB. This root-shoot ratio was estimated to 1. This is consistent with different values found in other studies like the ones found by Weigelt *et al*. 2005 (root-shoot ratio: 0.4 to 1.5) and also with field data from high altitude grasslands in Lautaret in the French Alps (VITAL project, 2010) (root-shoot ratio: 0.64 +- 0.23 for the roots within the 10 first cm of soil).

<u>Step 2: From BGB to BGB carbon stocks</u>

Carbon concentration in BGB was estimated from field data from high altitude grasslands in Lautaret in the French Alps (VITAL project, 2010) (%C BGB = 43.20 +- 1.79). This value is consistent with the ones found by Birouste *et al.* 2011 (%C BGB = 46.77 +- 2.11) for eighteen herbaceous species representative of plant communities from French Mediterranean succession. VITAL's value is however smaller than the one found by Silver & Miya 2001 (%C BGB = 59.40) but this may be linked to the fact that the latter was obtained from a review of root data across a wide range of latitudes and biomes.

D.2 Forests

<u>Step 1: From inventory statistics to harvestable volumes</u>

We used publicly available data from the Nation Forest Inventory (IFN www.ifn.fr) that gave per département the volume (m^3) and the surface (ha) of each forest type (broadleaves, conifers, mixt dominated by broadleaves and mixt dominated by conifers). Thus, IFN volume (m^3/ha) was obtained by Equation 3.

Equation 3

$$VI_k = \frac{VP_k}{SP_k}$$

With:

- k = forest type = broadleaves (f), conifers (c), mixt (m) dominated by broadleaves (m,f) and mixt dominated by conifers (m,c)
- VI = IFN volume = trunk volume (m3/ha)
- VP = forest type volume per department (m^3)

- SP = forest type surface per department (ha)

Step 2: Determining global conversion factors per département
Conversion factors (FC) transform IFN volumes to carbon stocks in biomass (both above and below-ground biomass). They are synthetic values that take into account branches and roots expansion factors, wood density and wood carbon rate. Carbofor project (Lousteau 2004) gave specific national conversion factors that rated for French forests: 0.535 for broadleaves and 0.361 for conifers.

Moreover, Carbofor project (Lousteau 2004) provided carbon stocks and IFN volume per département, which allowed calculating a global conversion factor per département (Equation 4). This global factor did not distinguish broadleaves from conifers.

Equation 4

$$FC_{g,d} = \frac{ST_d}{VI_d}$$

With:

- d = département
- VI = IFN volume = trunk volume (m^3/ha)
- ST = carbon stocks in wood biomass (tC/ha)

Step 3: Determining specific conversion factors per forest type and département
We deduced from global conversion factors two specific conversion factors per département, one for broadleaves and another for conifers.

Let A be the national conversion factors ratio constant (Equation 5).

Equation 5

$$A = \frac{FC_{France,f}}{FC_{France,c}} = \frac{0.535}{0.361} = 1.48$$

We made two hypotheses in order to assess specific conversion factors per département.

- First, for each département, the global conversion factor is equal to conifers and broadleaves conversion factors weighted by their proportion in volume (Equation 6).

Equation 6

$$FC_{g,d} = FC_{f,d} * P_{f,d} + FC_{c,d} * P_{c,d}$$

With:

- d = département
- FC = conversion factor (tC/IFNm^3)
- P = tree types proportion in volume, per département
- *Forest type* = global (g), broadleaves (f) or conifers (c)

- Second, after discussion with forestry experts, we choose a multiplicative hypothesis for the link between conversion factors at national and départemental scales (Equation 7).

Equation 7

$$\frac{FC_{f,d}}{FC_{c,d}} = \frac{FC_f}{FC_d} = A$$

With:
- *d* = département
- *FC* = conversion factor (tC/IFNm^3)
- *A* = national constant of conversion factors ratio
  Forest type = broadleaves (f) or conifers (c)

The resolution of Equation 6 and 7 gives us an expression of specific conversion factors per département (Equations 8 for broadleaves and 9 for conifers).

Equation 8

$$FC_{f,d} = A * FC_{c,d}$$

Equation 9

$$FC_{c,d} = \frac{FC_{g,d}}{P_c + A * P_f}$$

With:
- *d* = département
- *FC* = conversion factor (tC/IFNm^3)
- *A* = national constant of conversion factors ratio
- *Forest type* = broadleaves (f) or conifers (c)

No specific conversion factor for mixt forest types was directly available, thus we estimated it by weighting broadleaves and conifers conversion factors by the proportion in volume of mixt forest types dominated by broadleaves (m,f) and conifers (m,c), per département (Equation 10).

Equation 10

$$FC_{m,d} = FC_{f,d} * \frac{VP_{m,f}}{VP_m} + FC_{c,d} * \frac{VP_{m,c}}{VP_m}$$

With:
- *d* = département
- *FC* = conversion factor (tC/IFNm^3)
- *VP* = forest type volume per department (m^3)
- *Forest type* = mixt (m), broadleaves (f) or conifers (c)

Step 3: From harvestable volumes to carbon stocks in biomass
From IFN volumes and specific conversion factors per département, we obtained carbon stocks in biomass per département and forest types (tC/ha) (Equation 11).

Equation 11

$$ST_f = VI_f * FC_{f,d}$$
$$ST_c = VI_c * FC_{c,d}$$
$$ST_m = VI_m * FC_{m,d}$$

With:

- *ST* = carbon stock in biomass (tC/ha)
- *VI* = IFN volume (m^3/ha)
- *FC* = specific conversion factor (tC/IFNm^3)
- *Forest type* = broadleaves (f), conifers (c), mixt (m)

Step 4: Litter estimates

For dead organic matter stocks (litter), we used robust national estimates (Dupouey *et al.* 1999), as no data was available specifically for eco-regions of the French Alps.

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D. Supporting Information S2

For all following tables in Supporting Information S2, ecological parameters were abbreviated as follow:

Ecological parameter	Abbreviation
Agricultural production	crop
Forage production	fodd
Wood production	wood
Hydro-energy potential	hydro
Recreation potential	recre
Tourism	tour
Leisure hunting	hunt
Protected plant species	protp
Protected vertebrate species	protv
Erosion mitigation	eros
Protection against falling rocks	rock
Chemical water quality regulation	wql
Physical water quantity regulation	wqt
Biological control of pests	cbiol
Pollination	poll
Carbon storage	csto
Plant diversity	plant
Vertebrate diversity	vert

1. S2.A Pearson correlation coefficients between ecological parameters

Table S2.A: Pearson correlation coefficients (r) between ecological parameters. All results are significant at 5%, except non-significant ones labelled “n.s.”. Values in bold represent the top 15% (≥0.28) and are those detailed in the Results section.

r	**crop**	**fodd**	**wood**	**hydro**	**recre**	**tour**	**hunt**	**protp**	**protv**	**eros**	**rock**	**wql**	**wqt**	**cbiol**	**poll**	**csto**	**plant**	**vert**
crop	1,00	-0,24	-0,12	-0,19	**-0,35**	-0,20	-0,16	-0,07	0,15	-0,23	-0,18	**0,35**	-0,09	0,26	-0,05	-0,23	**0,54**	0,06
fodd	-	1,00	0,16	0,20	0,08	0,11	-0,05	-0,03	0,08	0,09	n.s.	-0,09	0,19	n.s.	0,08	0,21	-0,21	0,15
wood	-	-	1,00	0,06	-0,06	0,21	0,28	**-0,29**	0,23	0,21	0,09	-0,05	**0,37**	0,22	0,07	**0,44**	-0,23	**0,41**
hydro	-	-	-	1,00	0,21	n.s.	-0,21	0,10	-0,08	-0,17	0,15	**0,32**	0,08	-0,24	-0,17	0,05	-0,23	-0,05
recre	-	-	-	-	1,00	0,22	-0,06	0,10	-0,07	0,02	0,18	-0,14	n.s.	-0,22	0,03	0,08	**-0,34**	-0,11
tour	-	-	-	-	-	1,00	0,15	-0,16	0,13	0,11	n.s.	-0,21	0,11	0,06	0,12	0,25	**-0,36**	0,15
hunt	-	-	-	-	-	-	1,00	-0,08	0,06	0,23	-0,03	-0,17	0,08	0,16	0,21	0,23	-0,10	0,20
protp	-	-	-	-	-	-	-	1,00	-0,26	-0,22	n.s.	-0,07	-0,20	**-0,29**	-0,09	**-0,28**	0,23	**-0,31**
protv	-	-	-	-	-	-	-	-	1,00	0,10	-0,07	0,12	0,12	**0,44**	0,06	0,15	-0,03	**0,40**
eros	-	-	-	-	-	-	-	-	-	1,00	-0,09	-0,06	0,22	0,17	0,15	**0,47**	-0,19	0,21
rock	-	-	-	-	-	-	-	-	-	-	1,00	-0,06	0,05	-0,10	-0,06	0,10	-0,17	n.s.
wql	-	-	-	-	-	-	-	-	-	-	-	1,00	0,02	0,15	-0,08	-0,11	**0,32**	0,06
wqt	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,09	0,07	**0,37**	-0,10	0,21
cbiol	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,15	0,13	0,08	**0,60**
poll	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,14	0,04	0,10
csto	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	**-0,39**	**0,32**
plant	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	-0,15
vert																		1,00

2. S2.B Pairwise overlap rates between ecological parameters

Table S2.B: Pairwise overlap (O) between ecological parameters binary datasets. O varies from 0 (no overlap) to 1 (all pixels from the smaller network overlap with the second network). Values in bold represent the top 15% (≥0.40) and are those detailed in the Results section.

O	crop	fodd	wood	hydro	recre	tour	hunt	protp	protv	eros	rock	wql	wqt	cbiol	poll	csto	plant	vert
crop	1,00	0,03	0,07	0,14	0,07	0,13	0,15	0,16	0,30	0,03	0,00	**0,49**	0,08	0,32	0,23	0,04	**0,76**	0,23
fodd	-	1,00	**0,43**	0,15	0,31	0,29	0,17	0,16	0,39	0,29	0,07	0,22	**0,43**	0,29	0,33	0,23	0,16	**0,41**
wood	-	-	1,00	0,35	**0,40**	0,32	0,25	0,15	0,32	0,19	**0,53**	0,24	**0,49**	0,19	0,16	**0,53**	0,07	0,37
hydro	-	-	-	1,00	0,31	0,09	0,15	0,30	0,17	0,10	0,15	**0,57**	0,28	0,17	0,22	0,27	0,27	0,22
recre	-	-	-	-	1,00	0,39	0,21	0,39	0,28	0,29	**0,67**	0,22	**0,42**	0,23	0,33	**0,40**	0,14	0,28
tour	-	-	-	-	-	1,00	0,24	0,19	0,33	0,28	0,21	0,15	0,33	0,27	0,28	0,33	0,13	0,30
hunt	-	-	-	-	-	-	1,00	0,30	0,28	0,37	0,15	0,24	0,24	0,33	0,36	0,24	0,30	0,27
protp	-	-	-	-	-	-	-	1,00	0,12	0,16	0,28	0,19	0,17	0,16	0,39	0,15	**0,44**	0,12
protv	-	-	-	-	-	-	-	-	1,00	0,34	0,16	0,35	0,30	**0,45**	0,21	0,30	0,23	**0,47**
eros	-	-	-	-	-	-	-	-	-	1,00	0,03	0,24	0,25	**0,41**	0,29	0,27	0,18	0,32
rock	-	-	-	-	-	-	-	-	-	-	1,00	0,18	**0,40**	0,10	0,17	**0,42**	0,01	0,25
wql	-	-	-	-	-	-	-	-	-	-	-	1,00	0,22	0,38	0,21	0,21	**0,42**	0,30
wqt	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,22	0,27	**0,42**	0,10	0,29
cbiol	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,24	0,20	0,33	**0,50**
poll	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,14	0,32	0,18
csto	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,05	0,31
plant	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00	0,18
vert	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1,00

3. S2.C Altitude and land cover proportions by clusters (SOM)

Table S2.C1: Area (km² and % of total) covered by altitudinal ranges by cluster.

Altitude (m a.s.l.)	Cluster 1		Cluster 2		Cluster 3		Cluster 4		Cluster 5	
	km²	%	km²	%	km²	%	km²	%	km²	%
0-500	10783	75	5603	39	901	6	128	5	281	4
500-1000	3059	21	5859	41	3960	28	573	21	40	1
1000-1500	458	3	2470	17	5159	37	866	31	54	1
1500-2000	62	0	280	2	3400	24	731	26	735	12
2000-2500	29	0	58	0	677	5	384	14	3030	48
2500-4500	0	0	0	0	3	0	83	3	2168	34
Total	14391	100	14270	100	14100	100	2765	100	6308	100

Table S2.C2: Area (km² and % of total) covered by land cover categories by cluster.

Land cover category	Cluster 1		Cluster 2		Cluster 3		Cluster 4		Cluster 5	
	km²	%	km²	%	km²	%	km²	%	km²	%
Artificial areas	2239	16	122	1	164	1	6	0	140	2
Agricultural areas	10178	71	922	6	443	3	19	1	58	1
Grasslands and pastures	242	2	941	7	4449	32	283	10	1462	23
Forests	1022	7	8928	63	7660	54	1616	58	352	6
Semi-natural open areas	234	2	2946	21	668	5	246	9	191	3
Open spaces with little or no vegetation	292	2	392	3	686	5	595	22	4040	64
Wetlands and waterbodies	184	1	19	0	30	0	0	0	65	1

Total	14391	100	14270	100	14100	100	2765	100	6308	100

4. S2.D Overlap rates between high value clusters

Table S2.D: Pairwise overlap (*O*) between high value clusters of land cover types and of ecosystem services, as detected for a window size of 3*3 km. *O* varies from 0 (no overlap) to 1 (all pixels from the smaller network overlap with the second network). Values in bold represent the top 15% (≥0.51) and are those detailed in the Results section.

O	Agricultural areas	Artificial areas	Forests	Grasslands and pastures	Open spaces with little or no vegetation	Semi natural open areas	Wetlands and waterbodies
crop	**0,89**	0,46	0,03	0,06	0,00	0,06	0,45
fodd	0,15	0,15	0,25	**0,65**	0,20	0,06	0,24
wood	0,05	0,16	**0,56**	0,35	0,11	0,08	0,27
hydro	0,27	0,31	0,32	**0,55**	**0,65**	0,14	0,49
recre	0,14	0,05	0,34	**0,54**	**0,75**	0,31	0,22
tour	0,14	0,07	0,37	0,32	0,26	0,42	0,21
hunt	0,19	0,28	0,49	0,14	0,05	0,43	0,27
protp	0,20	0,36	0,14	0,28	0,47	0,19	0,20
protv	0,40	0,33	0,36	0,26	0,05	0,36	0,48
eros	0,12	0,07	**0,61**	0,23	0,01	0,48	0,10
rock	0,01	0,04	0,28	0,24	**0,58**	0,08	0,05
wql	**0,72**	0,29	0,15	0,04	0,01	0,04	0,32
wqt	0,20	0,20	**0,54**	0,43	0,10	0,22	0,23
cbiol	**0,60**	0,49	0,37	0,21	0,03	**0,58**	**0,55**
poll	0,19	0,17	0,20	0,24	0,08	**0,71**	0,08
csto	0,04	0,12	**0,64**	0,50	0,17	0,23	0,21
plant	**0,73**	**0,68**	0,10	0,06	0,04	0,23	0,51
vert	0,33	0,30	**0,52**	0,36	0,06	0,41	0,49

5. S2.E Chi² test residuals – Land cover distributions by Combination

Table S2.E: Chi² test residuals for land cover type distribution, by Combination of landscape heterogeneity and ecosystem services gamma index, with the entire study area as null model. All p-values are < 0.01.

Land cover types	LL	LH	HL	HH
Artificial areas	2,75	-15,89	14,30	-3,20
Agricultural areas	22,67	-9,11	3,51	-9,77
Grasslands and pastures	-15,85	-1,35	-3,67	18,65
Forests	-20,71	31,14	-10,43	2,63
Semi natural open areas	-13,99	-14,58	17,16	11,72
Open spaces with little or no vegetation	35,55	-17,82	-4,14	-18,01
Wetlands and waterbodies	-9,11	-5,65	-4,61	-3,80

6. S2.F Chi² test residuals – Altitude distributions by Combination

Table S2.F: Chi² test residuals for altitude distribution, by Combination of landscape heterogeneity and ecosystem services gamma index, with the entire study area as null model. Residuals were used to detect major departure from null expectation. All p-values for the Chi² test were < 0.01.

Altitude (m.a.s.l)	LL	LH	HL	HH

0 - 500	24,08	-14,90	16,43	-21,74
500 - 1000	-25,39	6,97	-7,02	21,85
1000 - 1500	-27,45	25,55	-11,89	17,17
1500 - 2000	-18,66	2,78	-2,01	14,93
2000 - 2500	17,14	-9,17	9,79	-12,58
2500 - 4500	50,42	-16,49	-14,47	-29,99

7. S2.G Overlap rates between Combinations and ecological parameters

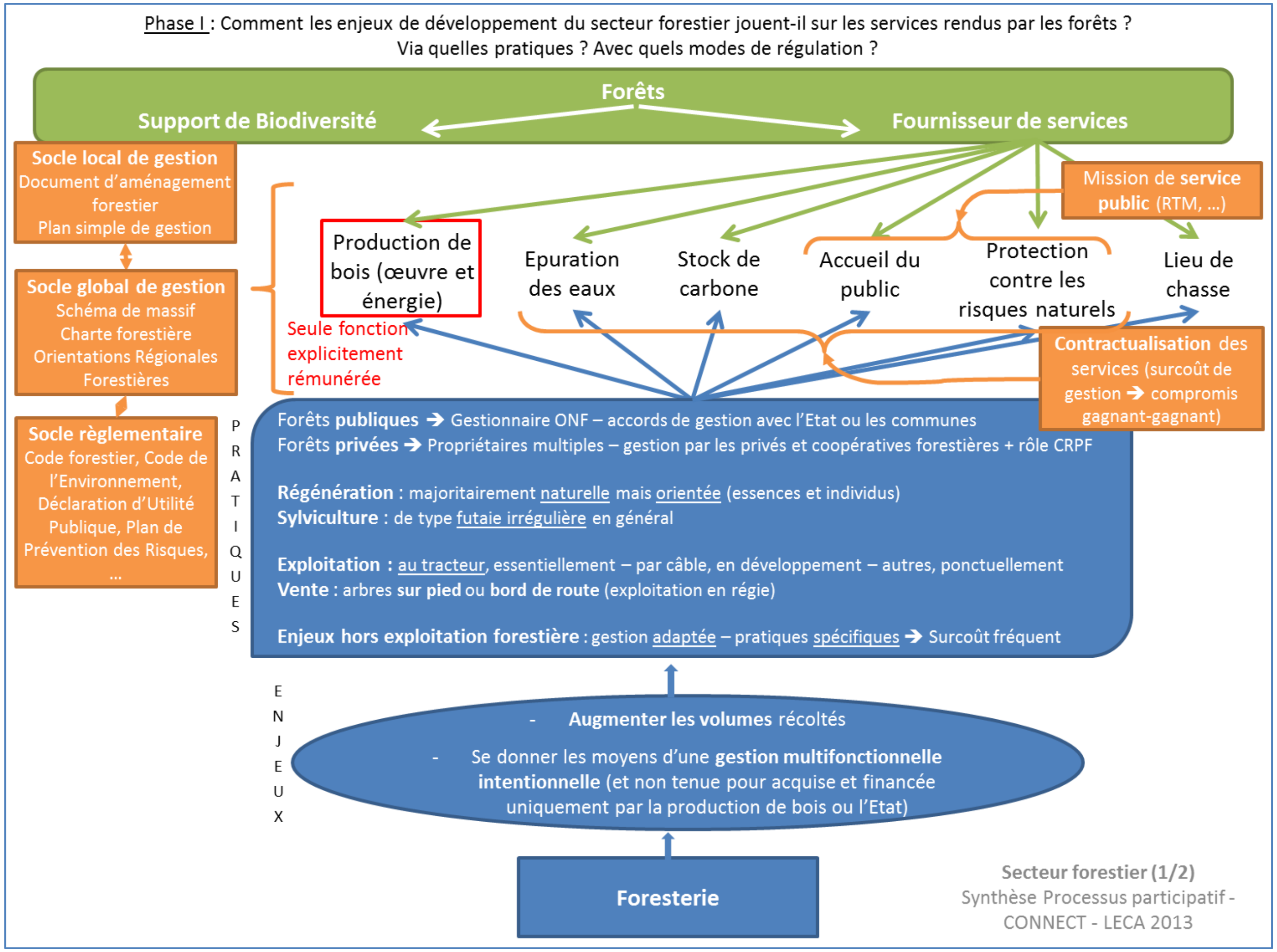
Table S2.G: Pairwise overlap (*O*) between Combinations and ecological parameters. *O* varies from 0 (no overlap) to 1 (all pixels from the smaller network overlap with the second network).

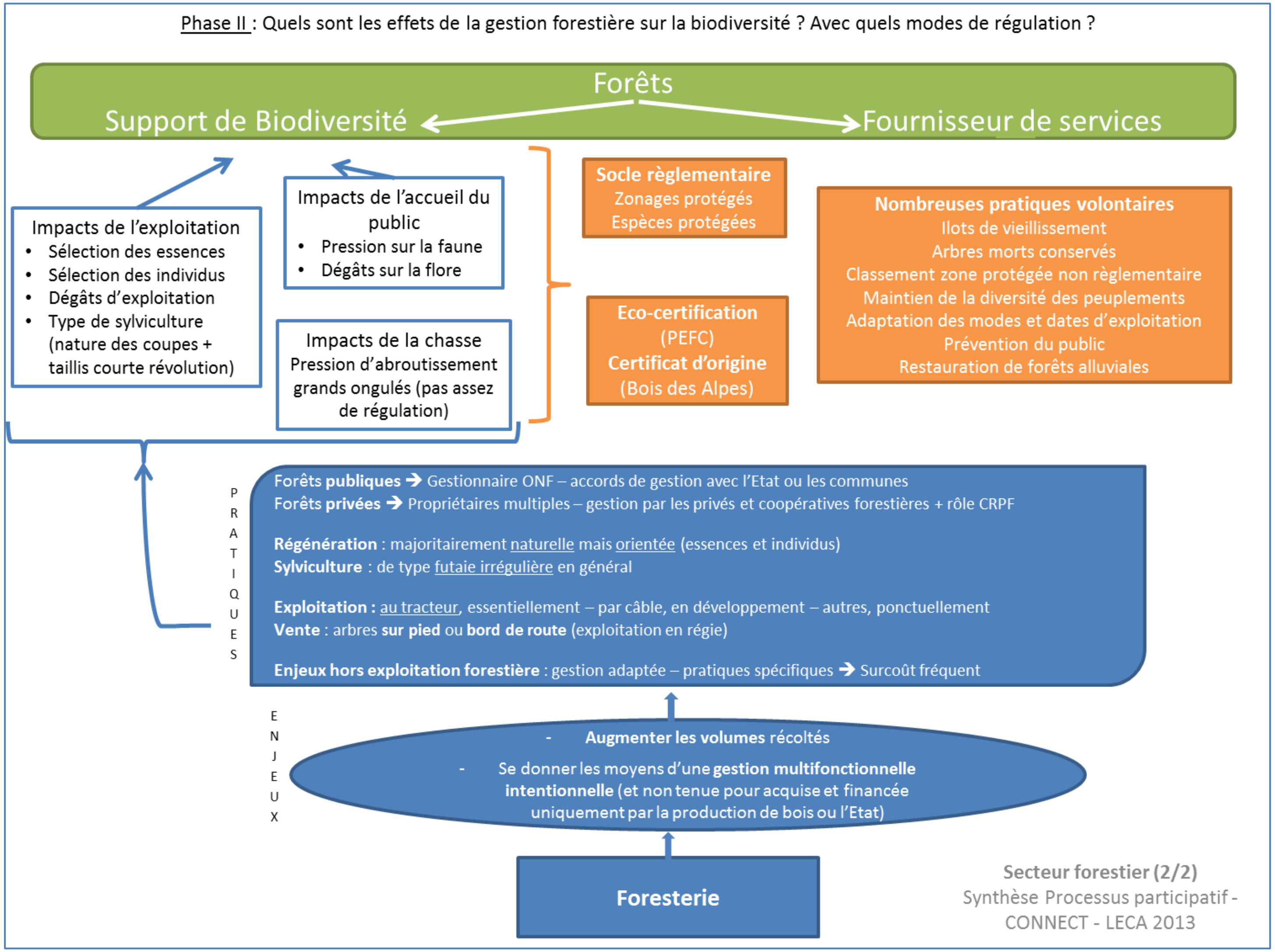
O	**LL**	**LH**	**HL**	**HH**
crop	0,38	0,15	0,16	0,29
fodd	0,06	0,20	0,10	0,63
wood	0,06	0,43	0,09	0,55
hydro	0,24	0,21	0,07	0,44
recre	0,31	0,40	0,23	0,42
tour	0,14	0,33	0,17	0,53
hunt	0,20	0,31	0,24	0,42
protp	0,31	0,15	0,22	0,32
protv	0,14	0,34	0,21	0,51
eros	0,21	0,29	0,26	0,40
rock	0,14	0,19	0,11	0,55
wql	0,21	0,32	0,16	0,46
wqt	0,11	0,42	0,12	0,50
cbiol	0,18	0,27	0,27	0,47
poll	0,24	0,22	0,33	0,41
csto	0,10	0,39	0,14	0,48
plant	0,40	0,18	0,34	0,27
vert	0,09	0,30	0,18	0,56

II. Appendices from Chapter II

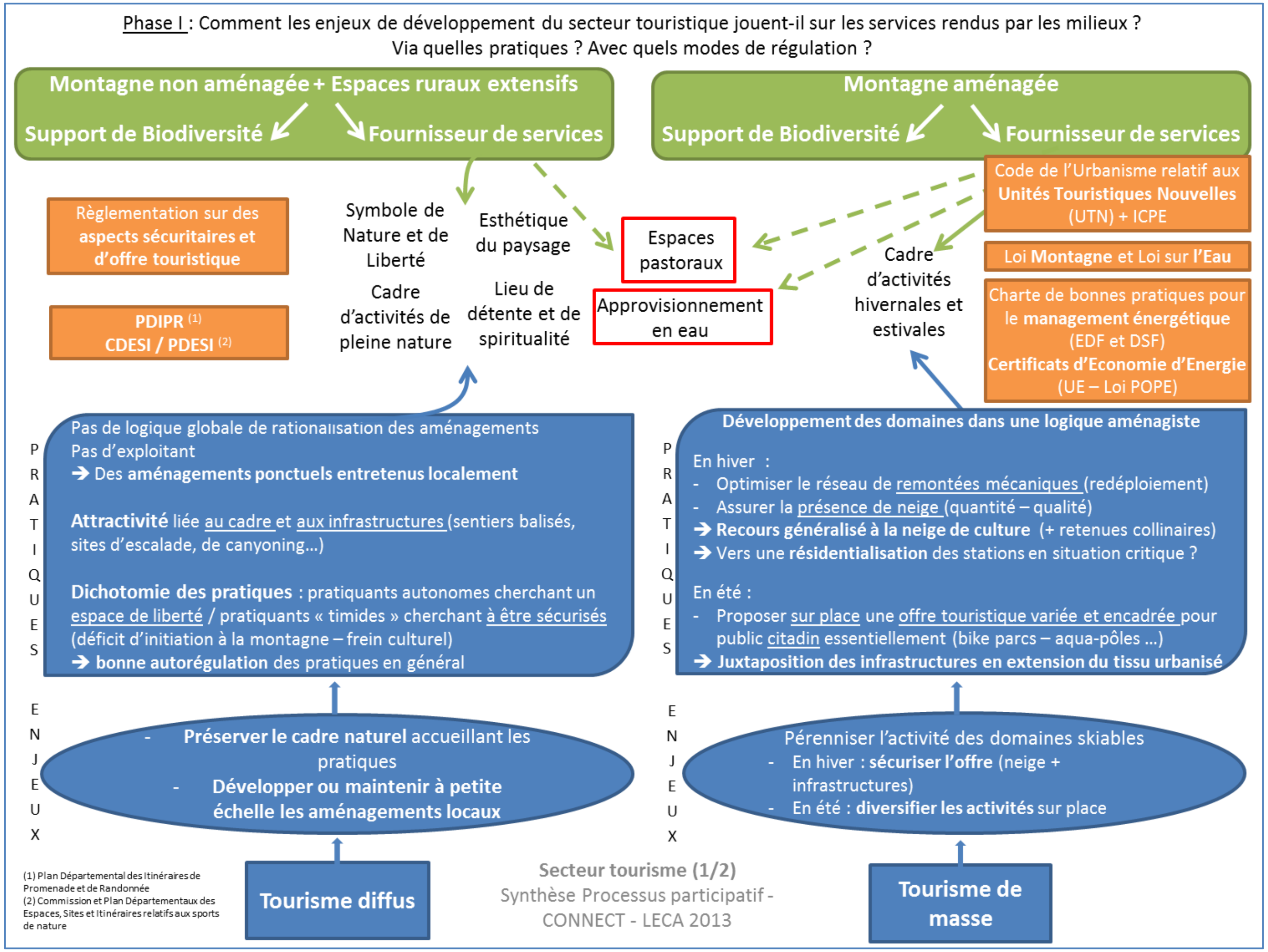
A. Sectoral syntheses

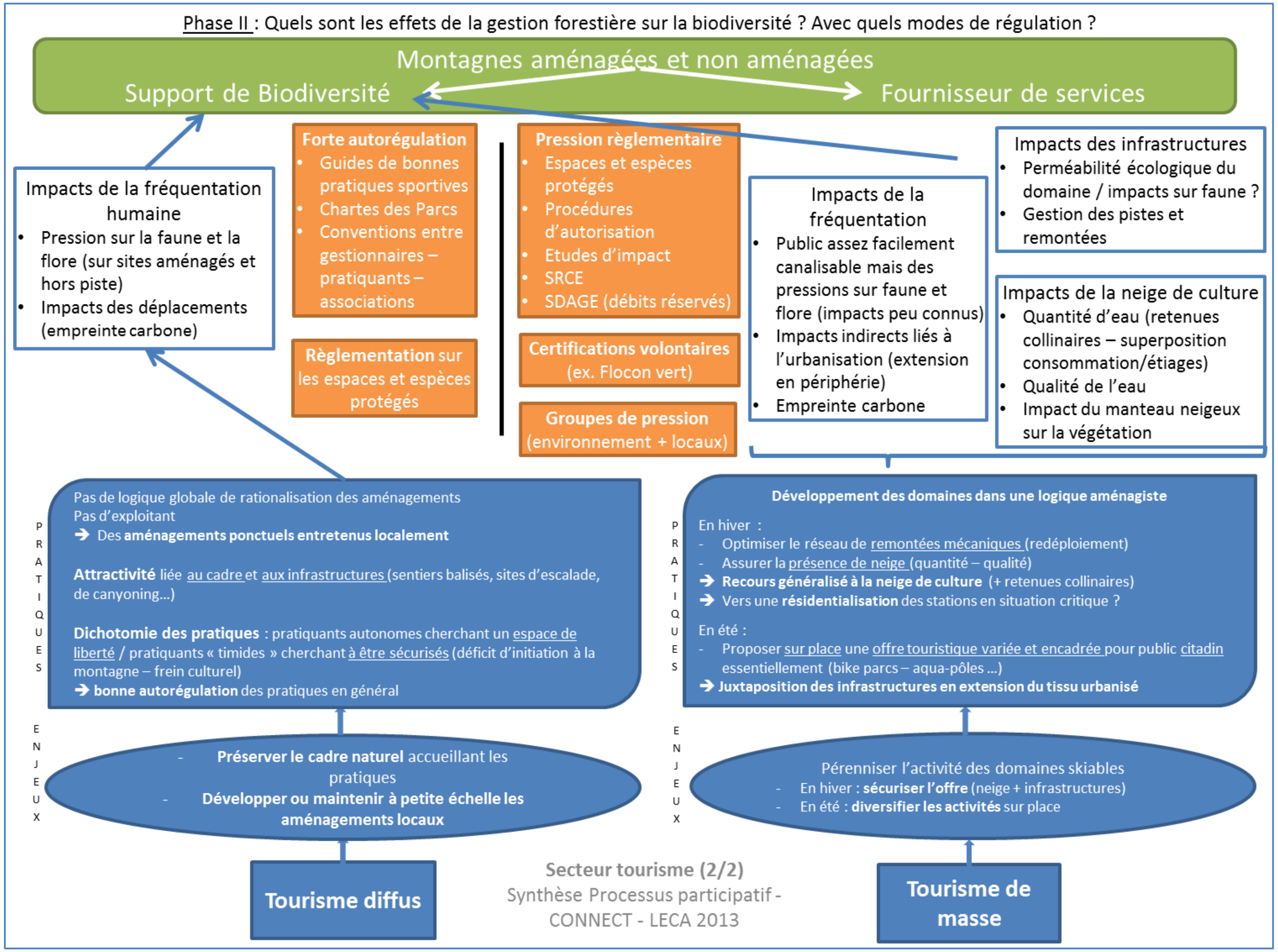
1. Forest sector




Phase II : Quels sont les effets de la gestion forestière sur la biodiversité ? Avec quels modes de régulation ?
Forêts
Support de Biodiversité
Fournisseur de services
Impacts de l'exploitation
• Sélection des essences
• Sélection des individus
• Dégâts d'exploitation
• Type de sylviculture (nature des coupes + taillis courte révolution)
Impacts de l'accueil du public
• Pression sur la faune
• Dégâts sur la flore
Impacts de la chasse
Pression d'abroutissement grands ongulés (pas assez de régulation)
Socle règlementaire
Zonages protégés
Espèces protégées
Eco-certification
(PEFC)
Certificat d'origine
(Bois des Alpes)
Nombreuses pratiques volontaires
Ilots de vieillissement
Arbres morts conservés
Classement zone protégée non règlementaire
Maintien de la diversité des peuplements
Adaptation des modes et dates d'exploitation
Prévention du public
Restauration de forêts alluviales
PRATIQUES
Forêts publiques → Gestionnaire ONF – accords de gestion avec l'Etat ou les communes
Forêts privées → Propriétaires multiples – gestion par les privés et coopératives forestières + rôle CRPF
Régénération : majoritairement naturelle mais orientée (essences et individus)
Sylviculture : de type futaie irrégulière en général
Exploitation : au tracteur, essentiellement – par câble, en développement – autres, ponctuellement
Vente : arbres sur pied ou bord de route (exploitation en régie)
Enjeux hors exploitation forestière : gestion adaptée – pratiques spécifiques → Surcoût fréquent
ENJEUX
- Augmenter les volumes récoltés
- Se donner les moyens d'une gestion multifonctionnelle intentionnelle (et non tenue pour acquise et financée uniquement par la production de bois ou l'Etat)
Foresterie
Secteur forestier (2/2)
Synthèse Processus participatif - CONNECT - LECA 2013

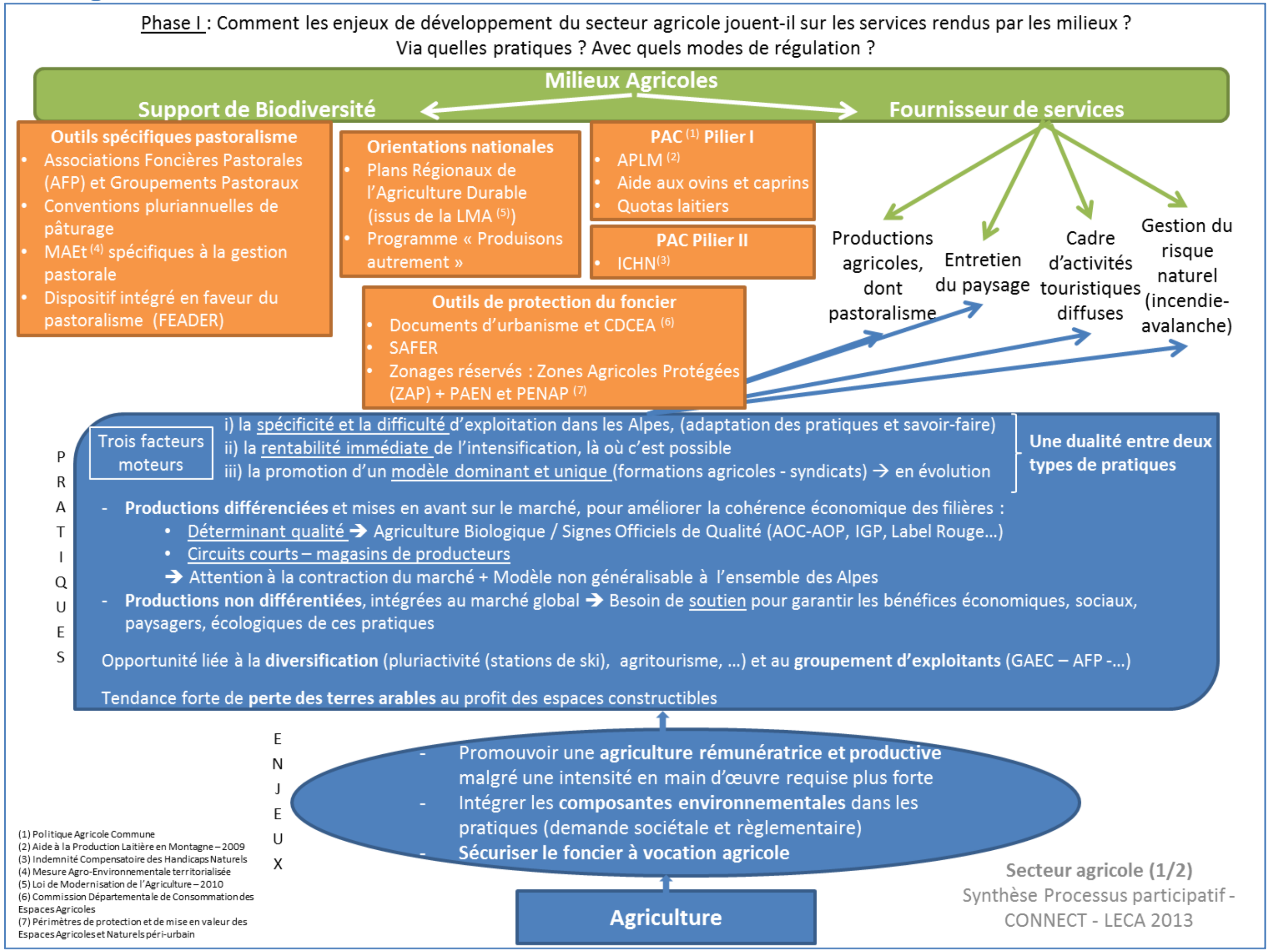
2. Tourism sector

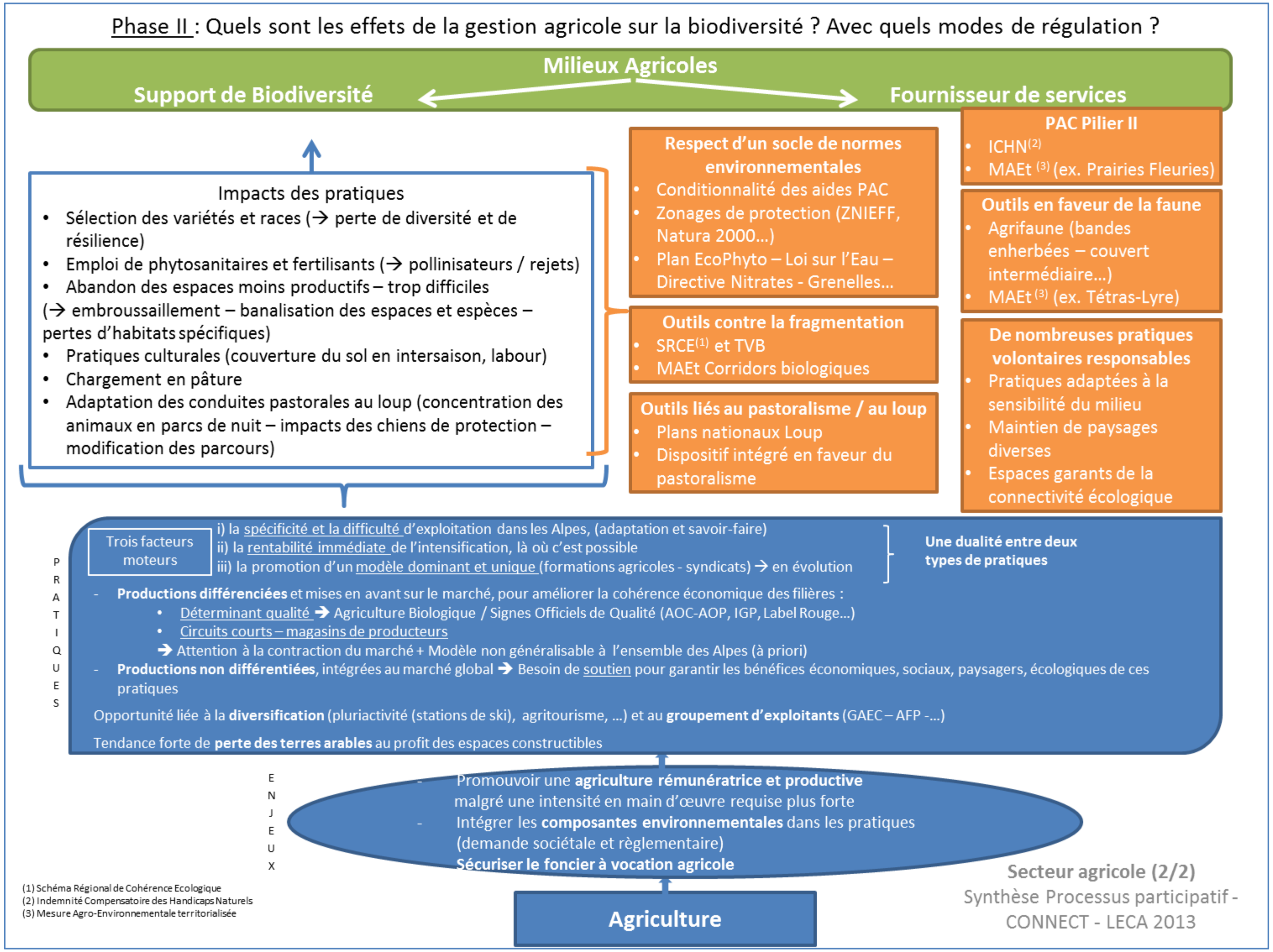




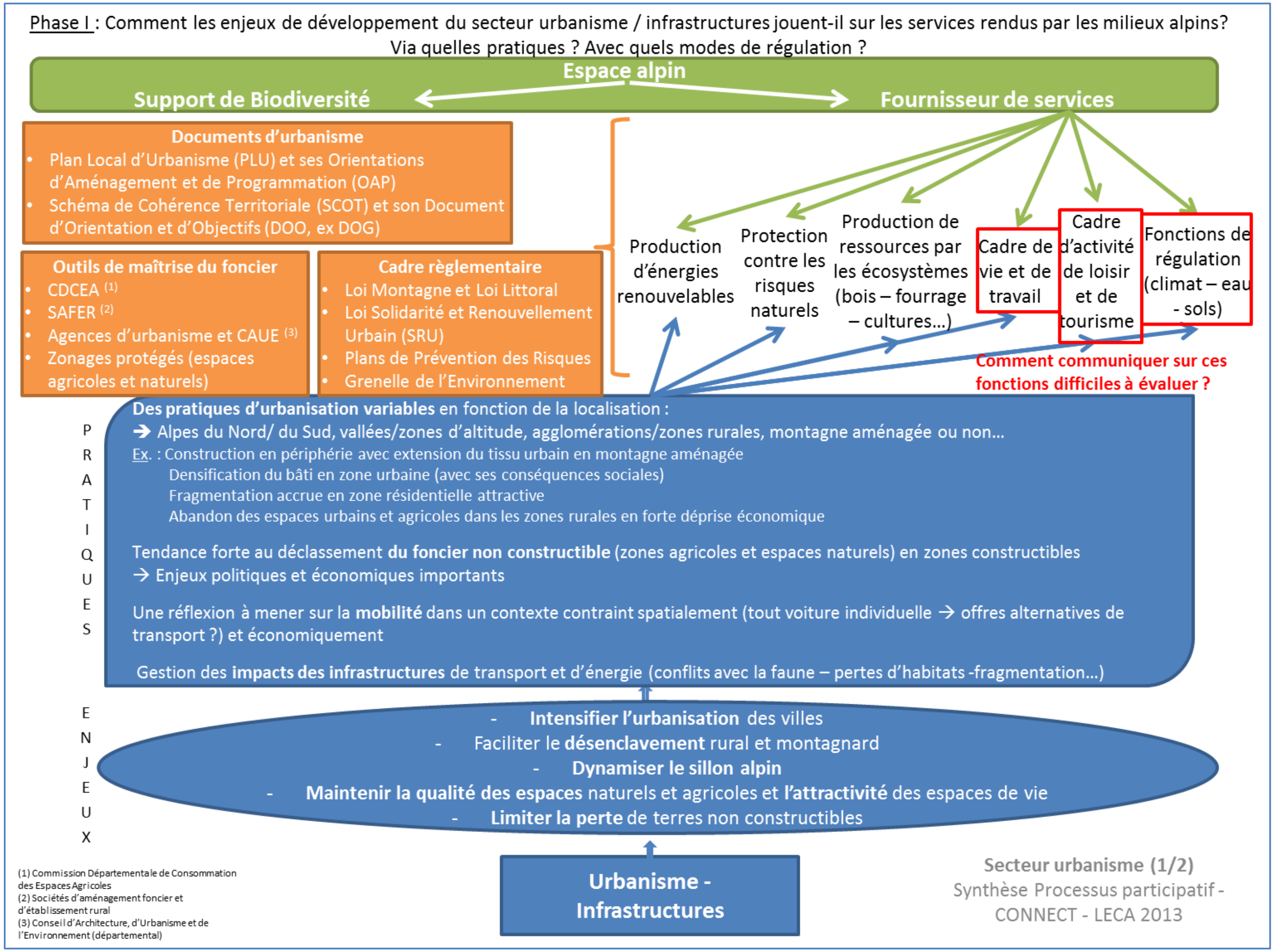
Phase II : Quels sont les effets de la gestion forestière sur la biodiversité ? Avec quels modes de régulation ?
Montagnes aménagées et non aménagées
Support de Biodiversité
Fournisseur de services
Impacts de la fréquentation humaine
Pression sur la faune et la flore (sur sites aménagés et hors piste)
Impacts des déplacements (empreinte carbone)
Forte autorégulation
Guides de bonnes pratiques sportives
Chartes des Parcs
Conventions entre gestionnaires – pratiquants – associations
Règlementation sur les espaces et espèces protégés
Pression règlementaire
Espaces et espèces protégés
Procédures d'autorisation
Etudes d'impact
SRCE
SDAGE (débits réservés)
Certifications volontaires (ex. Flocon vert)
Groupes de pression (environnement + locaux)
Impacts de la fréquentation
Public assez facilement canalisable mais des pressions sur faune et flore (impacts peu connus)
Impacts indirects liés à l'urbanisation (extension en périphérie)
Empreinte carbone
Impacts des infrastructures
Perméabilité écologique du domaine / impacts sur faune ?
Gestion des pistes et remontées
Impacts de la neige de culture
Quantité d'eau (retenues collinaires – superposition consommation/étiages)
Qualité de l'eau
Impact du manteau neigeux sur la végétation
PRATIQUES
Pas de logique globale de rationalisation des aménagements
Pas d'exploitant
→ Des aménagements ponctuels entretenus localement
Attractivité liée au cadre et aux infrastructures (sentiers balisés, sites d'escalade, de canyoning…)
Dichotomie des pratiques : pratiquants autonomes cherchant un espace de liberté / pratiquants « timides » cherchant à être sécurisés (déficit d'initiation à la montagne – frein culturel)
→ bonne autorégulation des pratiques en général
PRATIQUES
Développement des domaines dans une logique aménagiste
En hiver :
- Optimiser le réseau de remontées mécaniques (redéploiement)
- Assurer la présence de neige (quantité – qualité)
→ Recours généralisé à la neige de culture (+ retenues collinaires)
→ Vers une résidentialisation des stations en situation critique ?
En été :
- Proposer sur place une offre touristique variée et encadrée pour public citadin essentiellement (bike parcs – aqua-pôles …)
→ Juxtaposition des infrastructures en extension du tissu urbanisé
ENJEUX
- Préserver le cadre naturel accueillant les pratiques
- Développer ou maintenir à petite échelle les aménagements locaux
ENJEUX
Pérenniser l'activité des domaines skiables
- En hiver : sécuriser l'offre (neige + infrastructures)
- En été : diversifier les activités sur place
Tourisme diffus
Secteur tourisme (2/2)
Synthèse Processus participatif - CONNECT - LECA 2013
Tourisme de masse

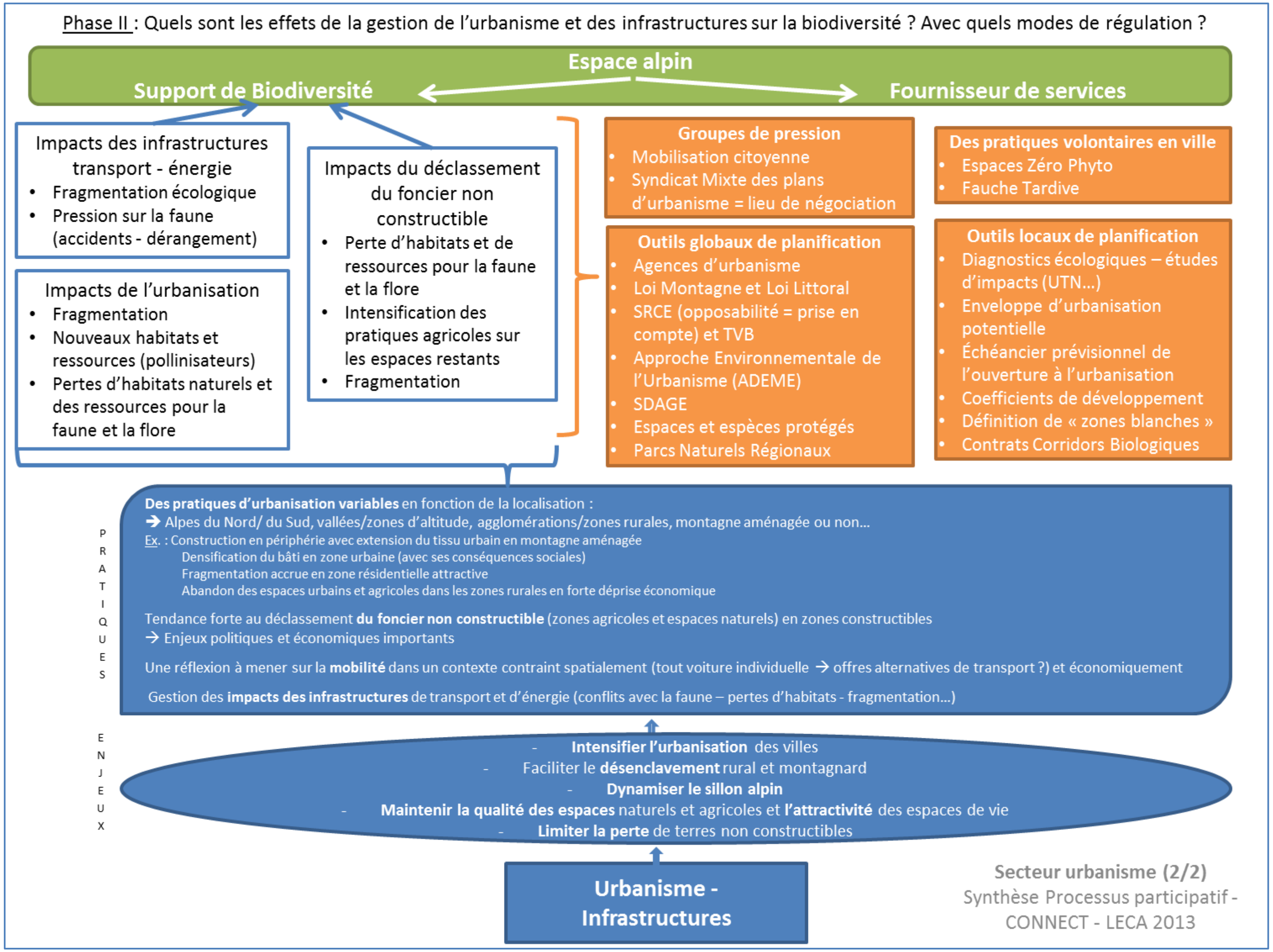
3. Agricultural sector




Phase II : Quels sont les effets de la gestion agricole sur la biodiversité ? Avec quels modes de régulation ?
Milieux Agricoles
Support de Biodiversité
Fournisseur de services
Impacts des pratiques
• Sélection des variétés et races (→ perte de diversité et de résilience)
• Emploi de phytosanitaires et fertilisants (→ pollinisateurs / rejets)
• Abandon des espaces moins productifs – trop difficiles
(→ embroussaillement – banalisation des espaces et espèces – pertes d'habitats spécifiques)
• Pratiques culturales (couverture du sol en intersaison, labour)
• Chargement en pâture
• Adaptation des conduites pastorales au loup (concentration des animaux en parcs de nuit – impacts des chiens de protection – modification des parcours)
Respect d'un socle de normes environnementales
• Conditionnalité des aides PAC
• Zonages de protection (ZNIEFF, Natura 2000…)
• Plan EcoPhyto – Loi sur l'Eau – Directive Nitrates - Grenelles…
Outils contre la fragmentation
• SRCE(1) et TVB
• MAEt Corridors biologiques
Outils liés au pastoralisme / au loup
• Plans nationaux Loup
• Dispositif intégré en faveur du pastoralisme
PAC Pilier II
• ICHN(2)
• MAEt (3) (ex. Prairies Fleuries)
Outils en faveur de la faune
• Agrifaune (bandes enherbées – couvert intermédiaire…)
• MAEt (3) (ex. Tétras-Lyre)
De nombreuses pratiques volontaires responsables
• Pratiques adaptées à la sensibilité du milieu
• Maintien de paysages diverses
• Espaces garants de la connectivité écologique
PRATIQUES
Trois facteurs moteurs
i) la spécificité et la difficulté d'exploitation dans les Alpes, (adaptation et savoir-faire)
ii) la rentabilité immédiate de l'intensification, là où c'est possible
iii) la promotion d'un modèle dominant et unique (formations agricoles - syndicats) → en évolution
Une dualité entre deux types de pratiques
- Productions différenciées et mises en avant sur le marché, pour améliorer la cohérence économique des filières :
• Déterminant qualité → Agriculture Biologique / Signes Officiels de Qualité (AOC-AOP, IGP, Label Rouge…)
• Circuits courts – magasins de producteurs
→ Attention à la contraction du marché + Modèle non généralisable à l'ensemble des Alpes (à priori)
- Productions non différentiées, intégrées au marché global → Besoin de soutien pour garantir les bénéfices économiques, sociaux, paysagers, écologiques de ces pratiques
Opportunité liée à la diversification (pluriactivité (stations de ski), agritourisme, …) et au groupement d'exploitants (GAEC – AFP -…)
Tendance forte de perte des terres arables au profit des espaces constructibles
ENJEUX
- Promouvoir une agriculture rémunératrice et productive malgré une intensité en main d'œuvre requise plus forte
- Intégrer les composantes environnementales dans les pratiques (demande sociétale et règlementaire)
Sécuriser le foncier à vocation agricole
Agriculture
(1) Schéma Régional de Cohérence Ecologique
(2) Indemnité Compensatoire des Handicaps Naturels
(3) Mesure Agro-Environnementale territorialisée
Secteur agricole (2/2)
Synthèse Processus participatif - CONNECT - LECA 2013

4. Urbanism and infrastructures




Phase II : Quels sont les effets de la gestion de l'urbanisme et des infrastructures sur la biodiversité ? Avec quels modes de régulation ?
Espace alpin
Support de Biodiversité
Fournisseur de services
Impacts des infrastructures transport - énergie
• Fragmentation écologique
• Pression sur la faune (accidents - dérangement)
Impacts de l'urbanisation
• Fragmentation
• Nouveaux habitats et ressources (pollinisateurs)
• Pertes d'habitats naturels et des ressources pour la faune et la flore
Impacts du déclassement du foncier non constructible
• Perte d'habitats et de ressources pour la faune et la flore
• Intensification des pratiques agricoles sur les espaces restants
• Fragmentation
Groupes de pression
• Mobilisation citoyenne
• Syndicat Mixte des plans d'urbanisme = lieu de négociation
Des pratiques volontaires en ville
• Espaces Zéro Phyto
• Fauche Tardive
Outils globaux de planification
• Agences d'urbanisme
• Loi Montagne et Loi Littoral
• SRCE (opposabilité = prise en compte) et TVB
• Approche Environnementale de l'Urbanisme (ADEME)
• SDAGE
• Espaces et espèces protégés
• Parcs Naturels Régionaux
Outils locaux de planification
• Diagnostics écologiques – études d'impacts (UTN…)
• Enveloppe d'urbanisation potentielle
• Échéancier prévisionnel de l'ouverture à l'urbanisation
• Coefficients de développement
• Définition de « zones blanches »
• Contrats Corridors Biologiques
PRATIQUES
Des pratiques d'urbanisation variables en fonction de la localisation :
→ Alpes du Nord/ du Sud, vallées/zones d'altitude, agglomérations/zones rurales, montagne aménagée ou non…
Ex. : Construction en périphérie avec extension du tissu urbain en montagne aménagée
Densification du bâti en zone urbaine (avec ses conséquences sociales)
Fragmentation accrue en zone résidentielle attractive
Abandon des espaces urbains et agricoles dans les zones rurales en forte déprise économique
Tendance forte au déclassement du foncier non constructible (zones agricoles et espaces naturels) en zones constructibles
→ Enjeux politiques et économiques importants
Une réflexion à mener sur la mobilité dans un contexte contraint spatialement (tout voiture individuelle → offres alternatives de transport ?) et économiquement
Gestion des impacts des infrastructures de transport et d'énergie (conflits avec la faune – pertes d'habitats - fragmentation…)
ENJEUX
- Intensifier l'urbanisation des villes
- Faciliter le désenclavement rural et montagnard
- Dynamiser le sillon alpin
- Maintenir la qualité des espaces naturels et agricoles et l'attractivité des espaces de vie
- Limiter la perte de terres non constructibles
Urbanisme - Infrastructures
Secteur urbanisme (2/2)
Synthèse Processus participatif - CONNECT - LECA 2013

B. Supporting Information

1. S1. Relations within ES potential supply facets

	ES_i	ES_i potential supply facet	ES_j	ES_j potential supply facet	Influence of external variable on ES_i	Influence of external variable on ES_j	Influence of ES_i on ES_j	Influence of ES_j on ES_i
1	Water quantities regulation	*Ecosystem ability to regulate the runoff regime in a river catchment, depending on vegetal cover and edaphic conditions*	Wood energy	*Logging residues from wood harvesting.*	Anthropic **water transfers** between watersheds modify the temporality and proportion of water fluxes and lessen ecosystems ability to regulate them.	-	-	-
2	Iconic species	*Abundance and richness of specific wild species*	Water quality	*Ecosystem ability to retain pollutants and nutrients from water fluxes, depending on vegetal cover and edaphic conditions*	**Urbanisation** induces a loss of resources and habitats for many iconic species.	Ecosystem disturbances caused by **urbanisation** decrease their ability to purify water.	-	-
3	Leisure hunting	*Presence of wild game species*	Wood production	*Biophysical potential to grow harvestable timber*	**Preservative hunting plans** increase game abundance.	-	Intense browsing by numerous wild ungulates damages young trees.	-
4	Fire risk mitigation	*Specific vegetation and land configuration reducing fire spread, notwithstanding human value and uses on the area*	Gravitational hazards mitigation	*Presence of natural protective elements from vegetal cover (forests - pastures) in areas exposed to gravitational risk but notwithstanding its human value and uses*	**Climate change** increases the risk of droughts and high temperatures, which limit the ability of ecosystems to mitigate fires.	**Climate change** accentuates risk of falling rocks in high altitude locations where low temperatures maintained rock stability in the past. Thus, the ability of	Ecosystems which are less prone to fire risk have more potential to protect against gravitational hazards than those weakened by fire.	-

						ecosystems to have protective properties overall decreases.		
5	Biological control of pests	*Presence of predators species*	Agricultural production	*Biophysical potential to grow harvestable agricultural products*	Pest predators can benefit from **protective status** which will reduce anthropic pressure (e.g. hunting).	-	Pest predators secure the ability of agricultural areas to produce biomass by limiting potential damages to crops, pastures etc.	Agricultural areas supply resources and habitats for pest predators.
6	Soil fertility	*Stock and recycling of nutrients needed for biomass growth, depending on above-ground biomass, soil biodiversity and edaphic conditions*	Erosion mitigation	*Soil retention and protection by plant cover, notwithstanding human value and uses of the area*	**Deep ploughing** impedes natural recycling of organic matter by soil fauna.	**Deep ploughing** turns under the residues of crop, letting bare soils more sensitive to erosion.	Fertile soils are constituted by stable aggregates which are less erodible.	Physical maintenance of soils is a prerequisite for maintenance of its fertility.

2. S2. Relations within ES demand facets

	ES_i	ES_i demand facet	ES_j	ES_j demand facet	Influence of external variable on ES_i	Influence of external variable on ES_j	Influence of ES_i on ES_j	Influence of ES_j on ES_i
1	Wood energy	*Accessible and profitable logging residues as renewable energy source*	Pollination	*Required pollination of agricultural areas (crops, orchards…) by wild pollinators*	**Renewable energies** are more demanded by society than by the past and this fosters the demand for wood energy.	-	-	-
2	Landscape aesthetics	*Satisfaction obtained from contemplating particular landscapes*	Gravitational hazards mitigation	*Protection of human activities and infrastructures*	**Increasing population** in the valleys increases the demand for nice landscape settings.	**Increasing population** in the valleys exposed to gravitational risks fosters demand for protection.	-	-
3	Nature tourism	*Accessible, secured and varied outdoor activities*	Agricultural productions	*Specific agricultural products*	**Higher overall summer temperatures** due to climate change will increase the demand for mountain activities in summer, due to higher attractiveness of fresh temperature.	-	An increased demand for nature tourism represents an economic opportunity for agricultural products, by increasing demand for alpine agricultural production.	-
4	Iconic species	*Social interest for designating iconic species, for instance large predators as wolves*	Agricultural productions	*Specific agricultural products*	The **increasing urban population** fosters the demand for “wild nature” with	The **increasing urban population** demands more food to be produced as the	Wolf presence appears much more important for some conservationists than preserved	-

					charismatic species, including wolves.	number of consumers increases.	agricultural productions, thus making demand decrease.	
5	Wood production	*Accessible and profitable timber*	Leisure hunting	*Accessible, undisturbed and numerous game*	Social demand for **local and natural building materials** fosters the demand for alpine wood production.	-	Forest managers and hunters conflict regarding wild ungulate abundance management.	Forest managers and hunters conflict regarding wild ungulate abundance management.
6	Nature tourism	*Accessible, secured and varied outdoor activities*	Leisure hunting	*Accessible, undisturbed and numerous game*	**Mountains** represent attractive features for tourists (verticality, climate, nature feelings, “mountain” water bodies…).	**Mountains** present attractive features for hunters (challenge, natural feelings) but also limit demand due to access difficulties (slopes, remoteness).	Local hunters and foreign tourists conflict regarding outdoor activities.	Local hunters and foreign tourists conflict regarding outdoor activities.

3. S3. Relations within ES actual supply facets

	ES_i	ES_i actual supply facet	ES_j	ES_j actual supply facet	Influence of external variable on ES_i	Influence of external variable on ES_j	Influence of ES_i on ES_j	Influence of ES_j on ES_i
1	Agricultural productions	*Crop and fodder yields*	Hydro energy	*Energy produced from hydroelectric plants*	Alpine societies inclination towards **agricultural employment** has decreased (low attractiveness and too many constraints). As a consequence, farms find few people willing to work in this sector and the actual production decreases in some sectors.	-	-	-
2	Wood production	*Amount of wood actually harvested in forests*	Agricultural productions	*Crop and fodder yields*	Increasing **land pressure** favours urban settlements at the expense of accessible forests supplying wood products.	Increasing **land pressure** favours urban settlements at the expense of fertile and favourable lands supplying agricultural products.	-	-
3	Wood production	*Amount of wood actually harvested in forests*	Fire risk mitigation	*Actual protection (or damage limitation) of human infrastructures from fire hazards*	Areas **protected by specific status** can limit wood harvesting in forests.	-	Forest harvesting roads limit fire spreading and ease firefighting.	-
4	Iconic species	*Actual designation of iconic species*	Wood production	*Amount of wood actually harvested in forests*	**Climate change** will affect negatively most	**Climate change** is anticipated to decrease	The presence of iconic protected species can limit	-

					iconic species, which are often already weakened.	abundance of noble harvested species.	forest area actually harvestable.	
5	Erosion mitigation	*Amount of soil erosion actually prevented by vegetal cover in managed and human-occupied areas*	Hydro energy	*Energy produced from hydroelectric plants*	**Soil artificialisation** induces increased runoff which decreases the actual erosion mitigation.	-	Limited sediments rates in hydroelectric infrastructures favour a good energy yield.	Hydro energy infrastructures induce sediment discontinuity which favours river depression in the rock and thus headward erosion (- on supply) but increase demand for the ES.
6	Maintain of water quality	*Amount of pollutants and nutrients actually retained and not reaching water bodies*	Fresh water supply	*Volume of water from the ecosystem actually used*	**Water policy** is intended to limit pollutants reaching water bodies.	Actual fresh water withdrawals are constrained by **policy** when water is a limited resource, and compromises must be obtained between users.	Ecosystems able to supply a water of good quality create diverse possibilities for water uses, and for a reduced treatment cost.	Water quality depends on volumes, and it is more difficult for ecosystems to purify water if the resource becomes scarce due to actual withdrawals.

4. S4. Relations between ES facets

Abbreviations stand for potential supply (P), demand (D) and actual supply (A) facets.

	ES_i	ES_i facet	ES_j	ES_j facet	Influence of external variable on ES_i	Influence of external variable on ES_j	Influence of ES_i on ES_j	Influence of ES_j on ES_i
1	Fresh water supply	*P - Freshwater available*	Pollination	*A - Amount of crops and cultures actually pollinated by wild insects*	**Climate change** will modify water fluxes and temporality.	-	-	-
2	Nature tourism	*D - Accessible, secured and varied outdoor activities*	Agricultural production	*P - Biophysical potential to grow harvestable agricultural products*	**Mountains** represent attractive features for tourists (verticality, climate, nature feelings, "mountain" water bodies…).	Specific climatic and altitudinal **conditions in mountains** limits biophysical potential to grow biomass	-	-
3	Water quantities regulation	*A - Actual regulation of water fluxes and stocks in soils*	Landscape aesthetics	*P - Potential landscape aesthetic quality*	Intensive **agricultural practises** (like high livestock density) favour soil compaction and limit soil water fluxes regulation.	-	Ecosystems not suffering from water excess or stress conserve an aspect which is positively perceived.	-
4	Leisure hunting	*D - Accessible, undisturbed and numerous game*	Iconic species	*P - Abundance and richness of specific wild species*	**Mountains** represent attractive features for hunters (challenge, natural feelings) but also limit demand due to access difficulties (slopes, remoteness).	Patrimonial species are well adapted to **mountain condition**s, and would get weakened by differing conditions.	Hunters tend to **manage ecosystem** (voluntary actions for maintaining open landscapes, for limiting invasive species dissemination…), which is favourable to	-

							many patrimonial species.	
5	Agricultural productions	*A - Crop and fodder yields*	Leisure hunting	*P - Presence of wild game animals*	A good **sector structuring and economic profitability** impacts positively agricultural production (short cycle – labels…).	-	Agricultural areas supply resources and habitats for many hunted species.	Competition between wild ungulates and livestock on pastures can make meadows yields and composition evolve.
6	Hydro energy	*A - Energy produced from hydroelectric plants*	Soil erosion mitigation	*P - Soil retention and protection by vegetal cover, notwithstanding human value and uses of the area*	**Climate change** will modify water fluxes and temporality (potential supply) and will increase demand for renewable local energy.	**Climate change** will induce more intense precipitations, which will decrease ecosystems ability to mitigate erosion.	Hydro energy infrastructures induce sediment discontinuity which favours river depression in the rock and thus headward erosion.	Limited sediments rates in hydroelectric infrastructures favour a good energy yield.

III. Appendices from Chapter III

A. Zupan et al. (subm.) The challenge of joint conservation planning for biodiversity's multiple facets and ecosystem service supply

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ABSTRACT

Much effort has been developed recently to assess trade-offs between ES and biodiversity in a context of conservation planning. However, most of the work has focused on species richness to target biodiversity ignoring other important facets, such as phylogenetic and functional, although they might better portray evolutionary history, ecological functioning and services than single count of species. We identified trade-offs between a set of ten ES, and between ES and biodiversity, the latter being captured by two indicators: the richness in vertebrate species (mammals, birds, amphibians and squamates) and the richness in evolutionary (ED) and functionally distinct (FD) species occurring in Europe. We observed in general little synergies between ES and between ES and biodiversity suggesting they should not be used as spatial surrogate for each other. With a spatial optimization procedure, we built alternative conservation scenarios and quantified how much a scenario based on ES maximization was able to capture ES and ED and FD species, and *vice versa*. The scenario targeting ES appeared to be more costly than the biodiversity scenario: to reach an average level of 10% representation of ES, only 3% more areas was needed in the biodiversity scenario compared to the ES one, while for the same level of species representation almost 6% more areas was needed in the ES scenario compared to the biodiversity scenario. The squamates appeared to be particularly affected for not being targeted with a loss of a third of their protection in the ES scenario compared to the biodiversity scenario when 50% of Europe was protected. The mapping of the win-win areas (areas of high priorities for both ES and biodiversity) showed that some areas of synergies between ES and biodiversity do exist but that there are no consistent bundles across Europe requiring careful guidelines for management. Preserving biodiversity together with the supply of ES is now explicitly required by global conservation policies. In that context, our work proposed an original way to provide a quantitative assessment of the trade-offs between ES and biodiversity, where the multiple facets of diversity is accounted for.

Keywords: conservation prioritization, ecosystem services, European vertebrates, evolutionary history, functional traits, spatial trade-offs and synergies.

1. INTRODUCTION

The Aichi targets (2010) set new objectives for global biodiversity conservation by incorporating ecosystem services (ES) – the benefit humans obtain from nature – into the nature protection and environmental management agendas. Based on the premise that biodiversity is the support of ecosystem services [1,2], targeting ES in conservation approaches has been proposed as a means of adding value to biodiversity conservation, hopefully achieving biodiversity targets while also safeguarding or enhancing human well-being. This approach implicitly assumes that ES are good proxies for each other and for biodiversity, and *vice versa* [3–5]. Yet, there is growing evidence for trade-offs both between different ES, and between ES and biodiversity. Therefore targeting simultaneously multiple ES and biodiversity in conservation poses several challenges and questions. First, enhancing the supply of multiple ES either locally or regionally is constrained by the basic fact that some ES are provided at the expense of others [6,7]. For example, a synergic relationship is often reported between carbon storage, soil retention and surface water quality in forests and extensively managed agricultural land, while an intensification of conventional crop production is known to compromise water quality [8–10]. Second, ensuring that the biodiversity traditionally targeted in conservation (e.g. rare, threatened or iconic species) is spatially congruent with the supply of ES conflicts with growing evidence that biodiversity and ES hotspots do not always overlap [11–13], consistent with the fact that ecosystem functioning and ES supply are often supported by the most abundant species, though not always [14]. Finally, biodiversity cannot only be represented by few iconic species while there is a growing awareness that evolutionary history and functional diversity are other facets of diversity that deserve attention [15]. However, safeguarding multiple biodiversity facets through different species groups (e.g. birds and mammals) may be difficult given evidence that taxonomic, phylogenetic and functional diversity for multiple taxonomic groups are not always spatially congruent [16–18].

Despite these three challenging lines of evidence, recent studies analysing the congruence between ES and biodiversity have mostly focused on a limited number of ES and have reduced biodiversity to species richness or vegetation cover [11,12,19,20]. While species richness is a natural biodiversity measure and a commonly used conservation currency [21,22], it ignores the phylogenetic or functional differences or similarities between species [16,17]. Yet, recent studies have shown that both phylogenetic and functional traits might bring additional and relevant information to species richness to predict ecological processes and services [23–26] and understand mechanisms of biodiversity patterns [27]. Also, the extinction of a species that belongs to an old lineage or to a specific functional group represented by very few species, would lead to a greater loss of evolutionary history or functional diversity than if it belongs to a species-rich young lineage or functional group [28–30]. Given that both evolutionary distinct (ED) and functionally distinct (FD) species provide complementary aspects of biodiversity [15], targeting them could, in theory, help future conservation actions by capturing the multi-faceted nature of biodiversity and representing the multiple dimensions of biodiversity that support ecosystem service provision. However, this issue has never been tested yet with empirical data.

In this paper, we tackle this challenge by analysing how maximizing the representation of ED and FD species on the top of species richness could also maintain a range of ES within a region, and *vice versa*. More specifically, we analysed how conservation strategies could best address these multiple challenges and associated dilemmas by designing conservation networks that best reconcile the preservation of multiple facets of biodiversity of several taxonomic groups along with the supply of multiple ES.

Our analyses focused on almost all vertebrate species of Europe (i.e. mammals, birds, amphibians and squamates) and a set of ten ES in the European Union (EU27). First, we analysed critical trade-offs and synergies between multiple facets of biodiversity of several taxonomic groups along with the supply of multiple ES by quantifying spatial co-variation between ES, between biodiversity indicators (i.e. total species richness and richness in ED and FD species for each taxonomic groups) and between biodiversity and ES. Second, we used alternative site-selection optimizations (scenarios) to evaluate how much a conservation scenario based on ES maximization performs to represent each ES and biodiversity, and *vice versa*. Finally, we mapped win-win areas, defined as locations where ES and biodiversity are maximised conjointly, – and described their representative bundles of ES and biodiversity.

2. METHODS

(a) Species distribution data

Distribution data for all vertebrates of the European Union were retrieved from [31]. Original data at 300m resolution was resampled at 10x10km resolution to match the resolution of the ecosystem services data. We kept the percentage of suitable 300m cells in each 10x10km pixel for each species to have a relative measure of coverage per species per pixel. In total we considered 160 mammals, 370 birds, 77 amphibians and 119 squamates for which we had the relevant traits and phylogenetic information and which represent 82% of the vertebrate species occurring within EU27.

(b) Functional traits.

The contribution of individual vertebrate species to ecosystem function is partly dependent on how species behave in their environment through their functional traits. We restricted our analyses to comparable traits between the four groups that represent different niche dimensions. These were body mass/body length, diet type, feeding behaviour, nesting position, reproduction and activity (see Table S1 for a description of the sub-classes of traits). These traits are known to relate to ecosystem functioning because they summarize or are linked to trophic interactions and resource acquisition and were selected for this reason [32]. We gathered all trait data from [15] (and references here in).

(c) Phylogenetic data

The phylogenetic trees for the four groups were gathered from [33]. All phylogenetic trees were dated molecular trees resolved at the species level available on TreeBase [33]. For each group, we used the maximum-likelihood tree from the 100 available on TreeBase to estimate the evolutionary distinctiveness of the species (see section below).

(d) Measure of functional and phylogenetic distinctiveness.

To measure both the evolutionary and functional distinctiveness (ED and FD, respectively), we built on the "evolutionary distinctiveness" measure developed in [30]. For a given species, the measure of distinctiveness equals to the sum of the branch length from the tip to the root of the tree divided by the number of species subtended to each branch (function *evol.distinct* in R package *picante*, [34,35]). This is applicable to phylogenetic trees but also to functional dendrograms (e.g. [15]). For each group of vertebrate, a functional dendrogram was built from the pairwise functional distances between species [36]. We used a mixed-variables coefficient of distance that generalizes Gower's coefficient of distance to allow for the treatment of various types of variables when calculating distances [37]. Euclidean distance was used for body mass/length (continuous traits) that were first log-transformed and normalized. We treated the remaining traits (categorical) with the Sorensen distance (coefficient of Gower and Legendre, [38,39]). A hierarchical clustering employing an average agglomeration method was then applied (UPGMA, function *hclust* in R package *stats*, [35,40]). To make ED and FD comparable between groups we standardized their values to the range between 0 and 1 by dividing all values by the maximum ED and FD, respectively.

(e) Ecosystem services mapping

We used ten different proxies for ES available at European scale on a 10x10km resolution (Table 1 and Text S1; [41]). Each proxy represents the capacity of ecosystems to provide services, also termed biophysical supply or potential [42,43]. Following the classification of the Common International Classification of Ecosystem Services [44], we included spatial proxies for two provisioning services (timber production and freshwater provision), five regulating services (air quality regulation, climate regulation, water regulation, water quality regulation, soil quality regulation, pollination and erosion control) and one cultural service (recreation). The values for each pixel were scaled between 0 and 1 and were further used in the prioritization exercise as conservation value to maximize.

Table 1: Ecosystem services (ES) and their associated indicators used in this study.

	Ecosystem services	Abbreviation	Indicators	Unit
Provisioning				
	Water provision	*wat prov*	Hydrological excess water (HXS)	mm / year
	Timber Production	*timb prod*	Stock	m3/ha
Regulation and maintenance				
	Climate regulation	*clim reg*	Carbon Storage	tonC/ha
	Water regulation	*wat reg*	Infiltration capacity	mm
	Water quality regulation	*wat qual reg*	Nitrogen retention capacity	%
	Soil quality regulation	*soil qual reg*	% Carbon	%
	Air quality regulation	*air qual reg*	Deposition velocity	cm/s
	Pollination	*pol*	Pollination capacity	Dimensionless
	Erosion control	*erosion cont*	Relative area of protective vegetation in risk zones	%
Cultural				
	Recreation	*recrea*	Recreation	Dimensionless

(f) Pairwise spatial co-variations between individual ES and biodiversity

We quantified the spatial co-variation between individual ES, between biodiversity indicators and individual ES and biodiversity indicators using spearman rank correlations within each of these sets of variables. The biodiversity indicators were the richness in vertebrate species (called vertebrate richness hereafter) per grid cell (all taxonomic groups included) and the richness of the top 10% evolutionary or functionally distinct species (referred as the top most EDFD species richness hereafter).

(g) Conservation scenarios

We conducted a series of conservation prioritizations (hereafter called conservation scenarios) where either the representation of ES or facets of biodiversity were maximized. We used the optimization software Zonation dedicated to spatial prioritization exercise for conservation planning [45,46]. The algorithm starts by calculating the conservation value of each cell of the region (here EU27) and then removes the least valuable cells iteratively while recalculating conservation values at each step. The input data to calculate the conservation values of each cell are spatial distribution data (either ES distribution or species distribution). Here, we used the "Core-area zonation" option as the removal rule, so that rare features (i.e. feature of small spatial extent) contribute more to the conservation value than broadly distributed features. Zonation also offer the option to weight particular ES or species according to the priority one want to give it in the optimization process. The output is a ranking of the entire region (i.e. EU27) from highest to lowest conservation priority [45,47]. We produced five alternative prioritization scenarios: one scenario where ES are maximized and four different biodiversity scenarios (see Table 2 for description). Since the prioritizations between the four biodiversity scenarios were highly correlated (Figure S1), only results relative to the *ES* and the *EDFD scenarios* are presented hereafter, the other ones are presented as supplementary material.

Table 2: Names of the different conservation scenarios, their associated conservation objectives, the spatial data used to calculate the conservation values of each cells of the EU27 grid and the weight applied to particular species/ES.

Name of the scenario	**Conservation objective**	**Spatial data used as conservation value**	**Weight**
ES scenario	Maximize the representation of all ES	spatial distribution of ES	None
Biodiversity scenario			
EDFD scenario	Maximize the representation of all species giving more weight to species that are evolutionary and/or functionally distinct	spatial distribution of vertebrate species	exp(ED) + exp(FD)
ED scenario	Maximize the representation of all species giving more weight to evolutionary distinct (ED) species	spatial distribution of vertebrate species	exp(ED)
FD scenario	Maximize the representation of all species giving more weight to functionally distinct (FD) species	spatial distribution of vertebrate species	exp(FD)
SP scenario	Maximize the representation of all species	spatial distribution of vertebrate species	None

(h) Evaluating and confronting the alternative conservation scenarios.

As a measure of representation of each feature in the different scenarios, we used the proportion of the range of each feature (either individual ES or species), at each iteration, that remains in the cells that have not been removed yet. This measure allows quantifying how much area is needed to achieve a given level of representation (e.g. how much protection of EU27 is needed to represent at least 10% of the range of all EDFD species).

For ES, we evaluated two levels of representation: the mean representation of all ES taken together, and the representation of each individual ES. The latter estimates trade-offs between ES within a given conservation scenario. For the *EDFD scenario*, we considered both the mean representation of the top most EDFD species (all taxonomic groups considered) and the mean representation of the top most EDFD species per taxonomic groups. The latter allows identifying trade-offs between taxonomic groups.

To assess the performance of the ES scenario at representing species, we recalculated the representation of each species while forcing Zonation to remove the cells in the same order as in the ES scenario. We did the same to assess the performance of the EDFD scenario at representing ES, this time recalculating the representation of each ES while forcing Zonation to remove the cells in the order of the EDFD scenario.

We finally assessed whether a given scenario (ES or EDFD, respectively) was better than random at protecting the alternate target (EDFD or ES, respectively), we calculated the mean feature's representation in a set of 100 random selection optimizations.

(i) Mapping conservation scenarios, win-win solutions and associated bundles of ES and biodiversity

To visualize the areas of highest conservation priority in the EU27, we mapped the spatial solutions (i.e. the rankings) arising from the site selection optimization for each alternative scenarios. In order to highlight win-win areas, we overlaid the ranking arising from both scenarios (the ES and the EDFD scenarios), and estimated the number of overlapping cells in each fraction of EU27 (i.e. top 1%, 5%, 10% etc. until reaching the full continent). To assess whether the overlapping cells in each fraction of EU27 where not picked by chance, we calculated for each fraction of protected EU27 the probability to pick twice the same cell under a binomial draw. In order to highlight which features were best represented in the overlapping cells with the highest score (i.e. cells that overlapped top 1% fraction of EU27 in both EDFD and ES scenario), we selected the overlapping cells and extracted the values of each feature to compare them to the respective mean values over Europe.

Finally, to describe the bundles of ES and biodiversity indicators associated with different win-win areas, we extracted for each cell from the 1% best fraction across the EU27, the value of each ES and each biodiversity indicator (vertebrate richness and the richness of the top 10% EDFD for each taxonomic group), and compared it to its mean value over Europe.

3. RESULTS

(a) Pairwise spatial co-variations between individual ES and biodiversity facets

Most of the pair-wise correlations between ES were positive but low ($r \leq 0.3$), while some pairs of ES showed a high positive correlation ($r \geq 0.5$, $p < 0.01$, Figure 1). This was the case for timber production and both climate regulation and soil quality regulation, for water provision and water regulation and for soil quality regulation and air quality regulation. Most of the pair-wise correlations between the different biodiversity indicators were close to zero, except for vertebrate richness that was positively and highly correlated to the richness of top 10% EDFD mammals and birds ($r > 0.6$, $p < 0.01$). Additionally, the different biodiversity indicators were in general not highly correlated to the different ES, except pollination that appeared to be strongly correlated to the richness of top 10% EDFD squamates ($r \approx 0.7$, $p < 0.01$). Interestingly, pollination was negatively correlated to both air and soil quality regulation ($r \leq -0.5$, $p < 0.01$) while it was not correlated to all other ES.

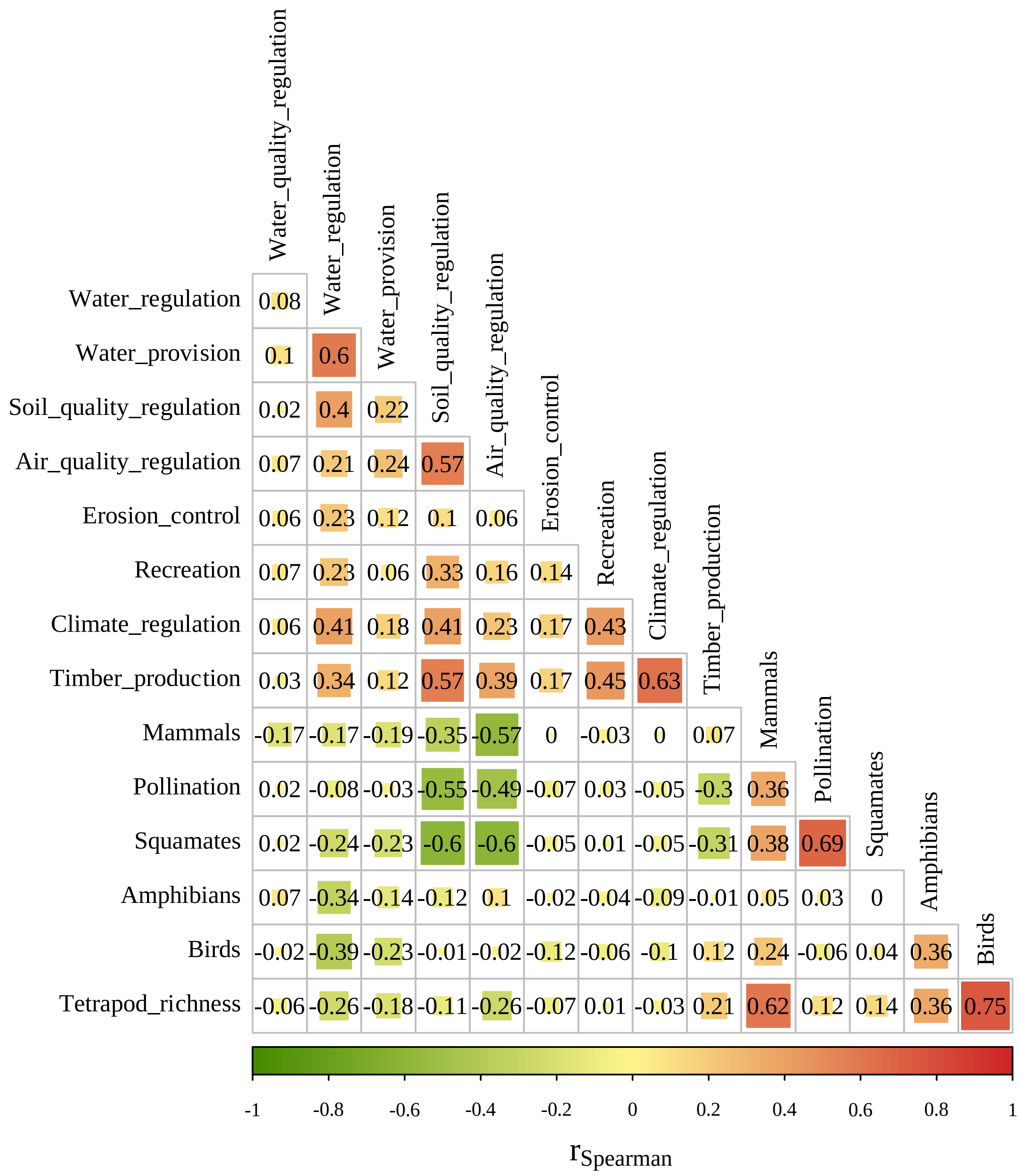

Figure 1: Pairwise correlation between ecosystem services and biodiversity indicators. Correlations are measured with the spearman rank correlation ($r_{Spearman}$) among individual ecosystem services (ES), among different indicators of biodiversity and between ES and the indicators of biodiversity. Green and red values correspond respectively to negative and positive correlation. The size of the square is proportional to the absolute value of the coefficient of correlation ($r_{Spearman}$).

(b) Evaluating and confronting the alternative conservation scenarios

When comparing the mean representation of ES and biodiversity in each alternative scenario, we found that both ES and the top most EDFD species were better represented in the ES and EDFD scenario respectively, both results departing significantly from random (Figure 2). Interestingly, both ES and EDFD were better protected than random in the scenario where there were not targeted. However, to reach a given level of representation in both scenarios, more protected areas were needed in the ES scenario than in the EDFD scenario. For example, to reach an average level of 10% representation of ES in the EDFD scenario, only 3% more areas was needed compared to the ES scenario, while to reach the same level of representation of EDFD species, almost 6% more areas was needed in the ES scenario compared to the biodiversity scenario (Figure 2 and Table S2).

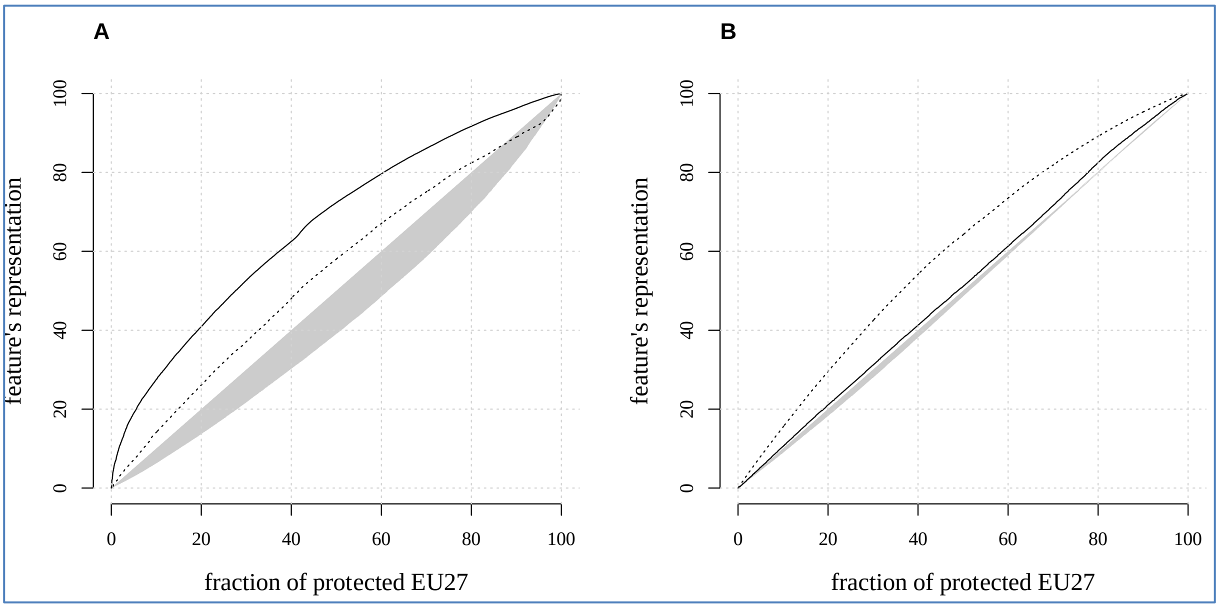

Figure 2: Representation of (A) top most EDFD species and (B) ES as total area selected for conservation increases (in %) in the ES scenario (dashed line) and in the EDFD scenario (plain line). Shaded areas indicate 95% confidence limits from 100 random prioritization runs. The x-axis (feature's representation) is the summed proportion of the distribution of the features (EDFD species or ES) remaining in each ranking fraction of Europe.

When analysed individually, the representation of each ES appeared to be uneven across scenarios (Figure 3 (C,D)), yet all of them were better represented in the ES scenario than under a random prioritization (all comparisons significant at $p < 0.001$, Table S4). Erosion control, climate regulation and soil quality regulation were on average the best-represented ES in the ES scenario with respectively 82, 72 and 69% of representation reached when 50% of the EU27 was protected. Comparatively, water provision and water and air quality regulation received the lowest representation at any fraction of the ranking with for example only 56, 52 and 59% of representation reached for within the 50% area priorities (Figure 3(C)). In contrast, when biodiversity was targeted (EDFD scenario), the representation of individual ES was found to be highly variable (Figure 3(D)), with pollination being better represented than any other ES - and even achieving higher levels of representation than with the ES scenario. For example, at 10% of EU27 protected, pollination reached a representation level of 64% in the ES scenario and of 71% in the EDFD scenario. Although erosion control was better represented than most other ES in the EDFD scenario, it was also the one with the sharpest representation decrease with proportion of area protected from the ES to the EDFD scenario (e.g. 28% at 50% of EU27 protected). The representation level of timber production and soil quality regulation also strongly decreased from the ES to the EDFD scenario (e.g. they lost 26 and 24 % respectively of their representation at 50% of EU27 protected).

When analysing taxonomic groups separately, we found that the top EDFD squamates and amphibians were on average better represented than the top most EDFD mammals and birds in the EDFD scenario, and that held true for the whole hierarchy of spatial priorities within Europe (i.e., within any priority fraction) (Figure 3(A)). For example, within the 10% area priorities, the representation of squamates was 22, 34 and 39% higher than the average representation of amphibians, mammals and birds respectively; with 50% of area prioritized the difference was of 18, 30 and 33% (Figure 3(A)). Obviously, the representation of the top most EDFD species was lower in the ES scenario compared to the scenario that targeted EDFD and that held true for any taxonomic group considered (Figure 3(B)). However, the decrease in representation of biodiversity from the EDFD to the ES scenario was uneven among taxonomic groups. For instance, within the 10% area priorities, the drop in representation for squamates was of 38% while it was lower for the other groups (13, 8 and 5% respectively for mammals, birds and amphibians) and of 32% at 50% when mammals, birds and amphibians loss 18, 8 and 4% of their representation respectively. This also translated in term of the proportion of additional areas needed in the ES scenario to reach the same level of species representation than in the EDFD scenario. Indeed, the number of additional areas needed to reach a representation level of 50% was higher for squamates (29%) than for the other groups (18, 8 and 3% for mammals, birds and amphibians respectively).

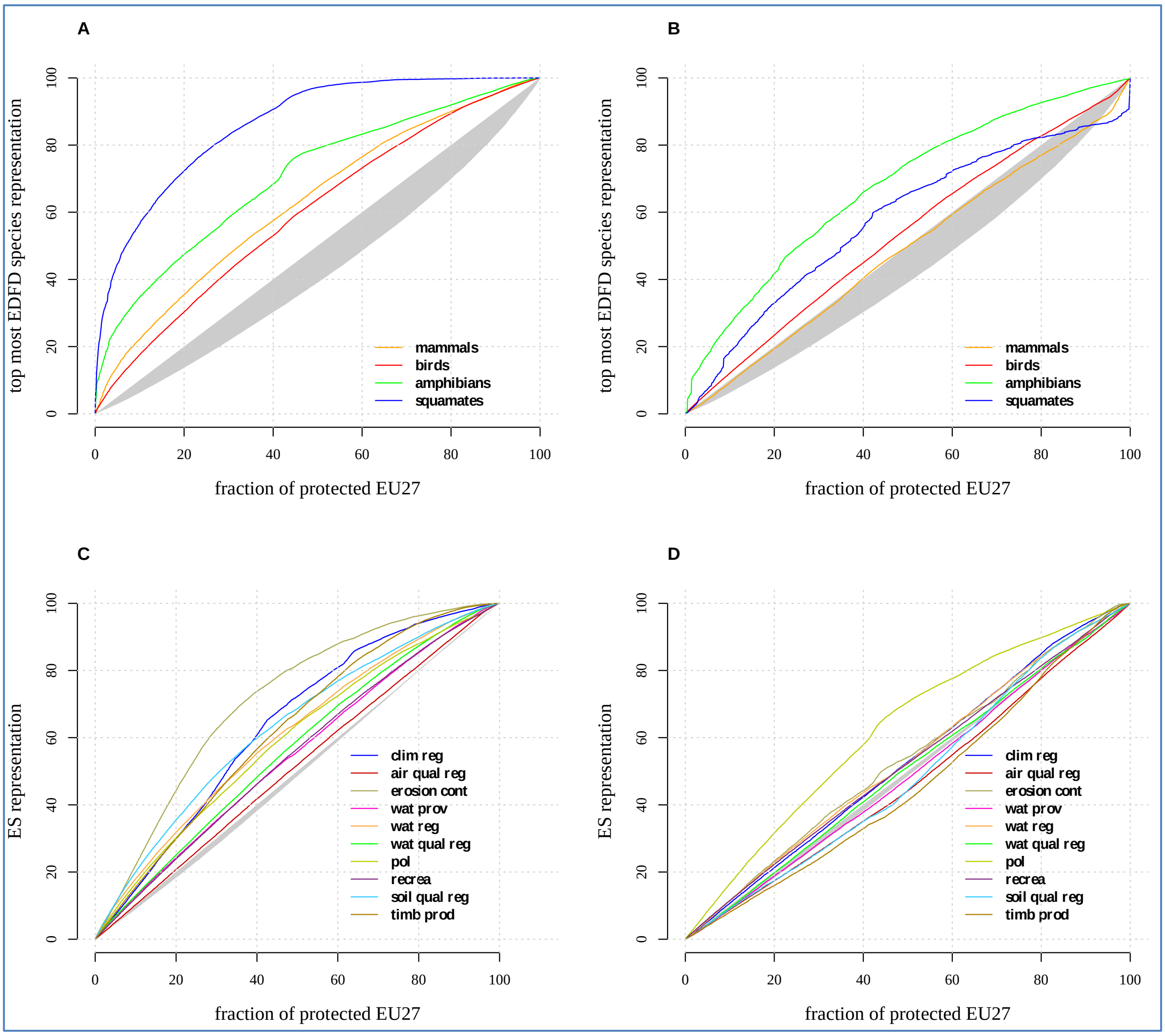

Figure 3: Representation of top most EDFD species per taxonomic groups in (A) the EDFD scenario and (B) in the ES scenario and the individual representation of ES in (C) the ES scenario (D) the EDFD scenario.

(b) Spatial pattern of priority areas

There were important differences across the two conservation scenarios (i.e. ES scenario vs. EDFD scenario, Figure 4). The ranking arising from the biodiversity scenario (i.e. the EDFD scenario, Figure 4A) showed three areas distributed along a latitudinal gradient. Southern regions (the Iberian, Italian and the Balkan Peninsula and the Mediterranean Islands) contained most of the top priority cells (red-orange areas) for species representation. Northern European countries such as the United Kingdom, Sweden and Finland also showed areas that were among the best fraction of the continent to represent vertebrate species. Comparatively, central Europe was ranked as least valuable for vertebrate species and its distinct species (dark blue areas on Figure 4(A)). In the ES scenarios Southern and Northern countries ranked high as well; however, the best top fractions were not always clustered in the same places as for the biodiversity scenario (Figure 4(B)). Ranking in central Europe was more heterogeneous, with small areas ranked as top fractions in Germany, Czech Republic and Austria and the Carpathians.

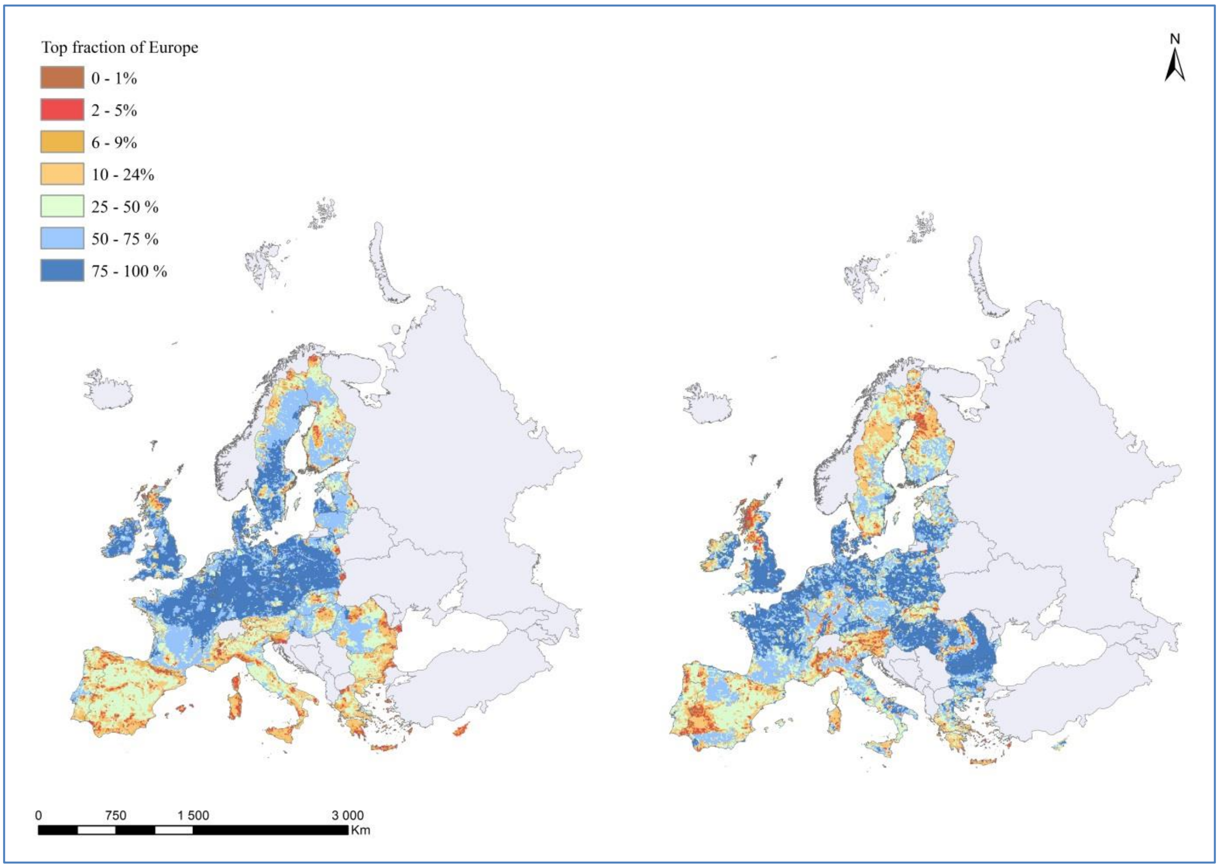

Figure 4: Maps representing the ranking of conservation priority in Europe according to (A) the EDFD scenario and (B) the ES scenario. The colours follow a gradient from red to blue with red areas depicting the most valuable fractions and the blue one representing the least valuable fraction of Europe.

(d) Win-win areas

Despite the apparent mismatch observed between the rankings of the two different conservation scenarios (Figure 4), a few areas shared high ranking across scenarios (red cells, Figure 5). For example, at 1% of protection in Europe, a significant number of cells overlapped (n=37, p<0.001, against an expected number of overlapping cells of 4.08, Table S5) between the scenarios. When examining which features were best represented in these overlapping cells, we observed distinct combinations of ES and biodiversity indicators in different regions (Figure 5, Table S6). For example, the highly ranked overlapping cells in Spain were characterized by high levels for pollination, recreation, climate regulation and top most EDFD amphibians and squamates. While also capturing high values for climate regulation, the Northern coast of Estonia was rather associated with air and soil quality regulation and the richness of top EDFD birds. In contrast, southern Slovenia had a high level of representation for erosion control, water provision and regulation, timber provision, and recreation together with top most EDFD mammal richness and total vertebrate richness.

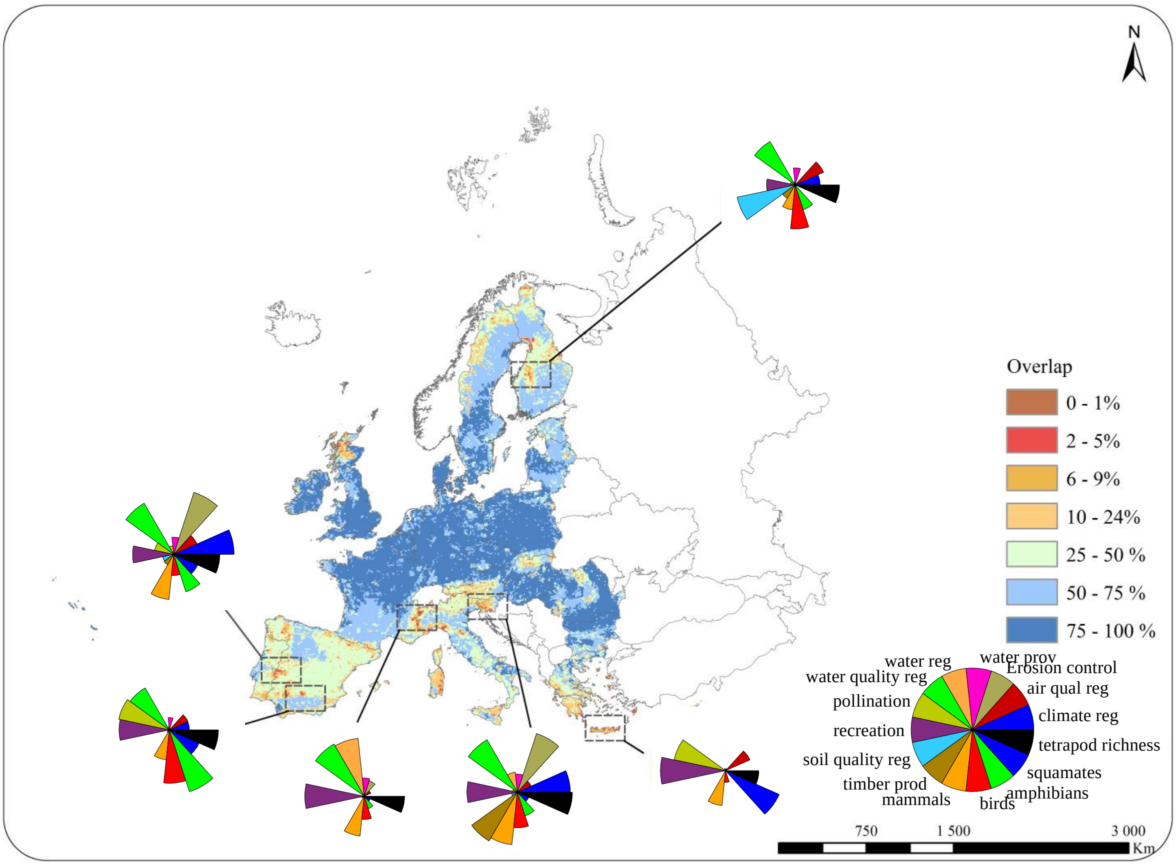

Figure 5: Win-win map between the ES scenario and the EDFD scenario. Colours follow a red to blue gradient with red cell corresponding to cells that overlap in the top fraction of both ES and EDFD ranking and blue cells being the ne that overlap in higher fraction (i.e. areas that are less valuable). The star plots represent examples of bundles of ES and biodiversity features that occur in the top overlapping cells.

4. DISCUSSION

Synergies and trade-offs among ES and among biodiversity indicators

While the spatial correlations between different pairs of ES were usually positive, they were also rather weak. This corroborates previous findings that individual ES are in general not good surrogates for each other [9,10,12,48]. When incorporated jointly in the same conservation scenario, not all ES were equally represented, some being favoured at the expense of others. In particular, air quality regulation, water provision and water quality regulation were under-represented in comparison to other services such as erosion control and soil quality regulation. Interestingly these inequalities were not identified with the simple correlation. For example, soil quality regulation was highly correlated to air quality regulation and we could have expected a more equal representation of both these ES in the ES scenario. These results might be partly explained by the prioritization scheme. Indeed, in each scenario we estimated the conservation value from the geographic range of the ES/species (total number of cells occupied by the ES/species out of the total number of cells of the region). This means that while the algorithm finds solutions that best retain the core areas of all ES/species, a larger fraction of the small-range ES/species will be retained as protected areas increase [45]. Erosion control and soil quality regulation are services that have smaller range than air quality regulation and water provision, which might explain why they are better represented especially in the first fraction of EU27 protected. These results also show that the conservation scenarios provide interesting and complementary insights by highlighting that co-variation of ES might not be influential when multiple ES are maximized jointly and supports the idea that ES interrelate in complex ways and that management for ES bundles is challenging [8,49–51].

Biodiversity indicators (represented here by vertebrate richness and the richness of top 10% evolutionary and functionally distinct species) showed positive but weak spatial co-variation. In particular, we did not detect a strong relationship among the distributions for the most distinct species (top most EDFD species) of the four different groups, suggesting that the distribution of the EDFD species of one taxonomic group is likely to be a weak predictor of the distribution of another group. This corroborates results from previous studies that have reported low congruency between different biodiversity facets [16,17,52] and/or different taxonomic groups [18,53,54]. Interestingly, EDFD squamates and amphibians were better represented than EDFD mammals and birds in the EDFD scenario at any fraction of EU27 protected. Like the good representation of erosion control and soil quality regulation in the ES scenario, squamates and amphibians are likely to be better represented than birds and mammals in the EDFD scenario because their ranges are in average much smaller in Europe.

Trade-offs and synergies between ES and biodiversity indicators

We observed that most individual ES were weakly or negatively correlated with biodiversity indicators, suggesting that in most cases the distribution of ES should not be expected to be a good surrogate for biodiversity and conversely. These weak or negative correlations between ES and biodiversity should be interpreted with care because our assessment is based on the spatial congruence between ES and biodiversity patterns modelled independently, and not on the biological and ecological mechanisms underpinning ES [2]. Such a weak biodiversity-ES relationship might reflect the dependence of the provision of some ES predominantly on biophysical factors rather than on abiotic factors (e.g. water provision), or the dependence of some other ES on species groups that were not incorporated in our analysis (e.g. plants for water quality regulation and trees for erosion control). In the latter case, we would expect co-occurrence of ES and biodiversity indicators only if these plants happened to provide habitat for diverse vertebrates.

Despite these mismatches, we showed that targeting biodiversity (or ES) allows a better representation of ES (or biodiversity) than under a random selection of sites. However, the protection of ES and biodiversity in the reciprocal scenarios (EDFD and ES scenario respectively) was generally not optimal. Indeed, when not targeted directly, the representation of species can drop dramatically. This is particularly true for EDFD squamates that lost about a third of their protection in the ES scenario compared to the EDFD scenario when 50% of Europe was protected. Knowing that squamates are undergoing global decline and are disproportionally vulnerable to anthropogenic pressure and climate change [55–57], basing future conservation strategies only on services-related criteria might have dramatic impact on such species group.

Similarly, the analysis of the representation of individual ES within the EDFD scenario showed contrasting results. Erosion control, timber production and soil quality regulation incurred more severe loss of protection within the EDFD scenario than other ES. Interestingly, pollination was very well protected in the EDFD scenario, and its representation was even better than in the ES scenario that directly targeted it along with other ES. This might partially be explained by the positive co-variation of squamates and pollination supply due to their co-occurrence in warmer regions (e.g. Mediterranean coast) favourable to both squamates and pollinating insects. However, this might also reveal that trade-offs between pollination and regulating services (e.g. soil and air quality regulation and timber production) led to its under-representation in the ES scenario. This potential trade-off might also reflect the way we estimated pollination, for which core forests are not considered while regulating services and timber production score high in forests [58].

Even if both scenarios performed less well than dedicated scenarios to protect non-targeted features, the biodiversity scenario appeared to perform better at protecting ES than the ES scenario was at capturing biodiversity. This means that a conservation strategy based solely on biodiversity criteria is more cost-effective to achieve given ES targets than a conservation plan based on ES would be to achieve a given biodiversity target. This has important implications because it means that a shift of conservation strategies toward the protection of ES only carries a high risk of losing biodiversity, while the converse risk for ES conservation would be lower with a traditional conservation strategy.

Given the trade-offs detected between some ES, the question of which ES should be maximized together with biodiversity should also be raised, as co-maximization of all ES is not feasible. For instance, alternative scenarios should be assessed to examine compatibility of conservation strategies of different categories of ES (provisioning, regulating, cultural) with biodiversity conservation. For instance, protection for regulation services might be more compatible with biodiversity protection than protection for provisioning services [20,59,60], as both regulation services and biodiversity rely on high quality natural habitat [13]. Also, our analysis highlights even conclusions among regulation services.

Win-win areas in Europe

Although we did not perform a specific cluster analysis to identify bundles comprising both ES and biodiversity facets [9,61], the mapping of win-win areas revealed strong heterogeneities across Europe. The identification of region-specific groups of best represented ES and EDFD species groups highlights a diversity of most valuable ES and biodiversity combinations, and thereby complementarities across the EU27 territory. These need to be considered for trans-national conservation and land planning. Also, dedicated management approaches would be needed for these individual bundles considering specific requirements for component species and ES.

Perspectives for conservation

Although our analysis made a significant progress on previous approaches, further issues will need to be addressed, such as scales of analysis in relation to scales of conservation and management planning. Adopting a continental approach to conservation planning by working at the European scale might be the most cost-effective in terms of area needed. However, conservation plans and policies are likely to be drawn at national or state scale. Likewise, the relevant scales for maximizing the provision of ES are hotly debated [62–66]. First, from an ecological perspective, it might make no sense to maximise some ES such as pollination at European scale given the short flying range of most pollinators that will require maximizing their local abundance. Second locations for supply and use differ across ES: some ES are provided locally and their consumption depends on the proximity of the ES to local population (e.g. water regulating service or soil erosion protection); other ES are enhanced locally (e.g. climate regulation) but their benefits (e.g. climate change mitigation) operate at global scale. Finally, trade-offs and synergies between biodiversity and ES are scale-dependent [59].

Conclusion

Our approach offered an evaluation of the compromise that conservationists will face when attempting for a synergic conservation of ES and biodiversity, which has become explicitly required by the European Union in its policy for nature conservation [67]. The European Union Biodiversity Strategy to 2020 includes multiple targets, among which protecting and restoring biodiversity and associated ecosystem services. Identifying priority areas for conservation, assessing current conservation measures, quantifying areas for ES restoration are key to reach 2020 goals. Our work exemplifies novel means to support this process by injecting new information on ways to assess the interdependence of ES and biodiversity, where a specific effort is made to quantify the multiple facets of diversity, and to provide quantitative information on conservation scenarios.

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6. SUPPORTING INFORMATION

Text S1. Supplement on ecosystem services

We used a European reference grid of 10 km (EEA, 2007) to map 10 ecosystem service indicators or proxies. We mapped indicators for the supply of ecosystem services. Each ecosystem service indicator represents therefore the potential or capacity of ecosystems present in each 10 km grid cell to deliver ecosystem services given suitable environmental conditions. Most ecosystem services maps were aggregated for this purpose to the desired resolution using zonal statistics, unless stated otherwise.

Timber production. Timber production services refer to the products from trees harvested from natural forests and plantations. The timber stock of each cell was estimated based on Gallaun et al. (2010) who combined national forest inventory data and remotely sensed data to produce pan-European maps on growing stock at 1 km resolution.

Fresh water supply. Freshwater provision accounts for the availability of fresh water from inland bodies of surface waters. We estimated the capacity of grid cells to provide a reserve of freshwater based on the hydrological excess water (HXS) in each cell. HXS is the difference between rainfall and evapo-transpiration (Wriedt and Bouraoui 2009).

Air quality regulation. This service refers to the influence of ecosystems on air quality by emitting chemicals to the atmosphere or by extracting chemicals from the atmosphere. We used the deposition velocity as an indicator for the capacity of vegetation in each grid cell to remove pollutants from the atmosphere (Nowak et al., 2006). The main ecosystem based parameters affecting deposition velocity are the height of the vegetation (related to the roughness length of the land) and the leaf area index (LAI). Both parameters are high in forests, thus explaining their substantial contribution to the provision of clean air. Average annual deposition velocities (cm s^{-1}) were calculated for NO_2 using the methodology applied by Pistocchi and Galmarini (2010).

Climate regulation. Climate services are defined as the influence that ecosystems have on the global climate by emitting greenhouse gasses to the atmosphere or by extracting carbon from the atmosphere. Carbon storage was used as a proxy to estimate the capacity of grid cells to contribute to climate change mitigation. Carbon storage data were derived from Gibbs (2006). This spatially-explicit global data set provides estimates and spatial distribution of the above- and below-ground carbon stored in living plant material in ton ha^{-1}. The data set was created by updating the classic study by Olson et al. (1985) with a map of global vegetation distribution, which is available at 1 km resolution (Global Land Cover database; GLC2000).

Water regulation. Water regulation refers to the influence ecosystems have on the timing and magnitude of water runoff, flooding and aquifer recharge, particularly in terms of water storage potential of the ecosystem. We used annually aggregated soil infiltration (mm) as an indicator for the capacity of terrestrial ecosystems to temporarily store surface water (Pistocchi et al., 2008). The data used are derived from the MAPPE model (Pistocchi et al. 2008; Pistocchi et al. 2010). MAPPE stands for Multimedia Assessment of Pollutant Pathways in the Environment of Europe and consists of models that simulate the pollutant pathways in air, soil sediments and surface and sea water at the European continental scale. Monthly infiltration of precipitated water in soils was calculated by distributing the net precipitation over run off and infiltration.

Water quality regulation. Water purification refers to the capacity of ecosystems to retain, process and remove pollutants, sediments and excess nutrients. Using nitrogen as common water quality indicator, Maes et al (2012) mapped nitrogen retention capacity as the proportion of nitrogen that is removed from rivers and lakes before it is discharged to a downstream catchment. Here we used the same mapping approach which is based on a pan-European statistical model developed to estimate total nitrogen fluxes to surface water in large river basins (Grizzetti et al., 2008).

Pollination. Pollination services are essential to maintain and enhance the production of crops that are dependent on insect pollination. We used the relative pollination potential map of Zulian et al. (2013) who developed a European wide model to map the relative capacity land pixels to provide pollination services to adjacent crops.

Erosion control. This service refers to the role of vegetation in soil conservation and in preventing the siltation of waterways and landslides. We combined a soil erosion risk map with a natural vegetation map to estimate the potential of ecosystems to help prevent erosion in risk areas. Erosion risk was assessed using K-factor (Panagos et al. 2012). Soils with values > 0.045 (t ha h)/(ha MJ mm) are considered sensitive to soil erosion. The final indicator is the relative surface area of natural vegetation on soils sensitive to erosion.

Maintenance of soil fertility. Soil services relate to the role ecosystems play in sustaining soil biological activity, diversity and productivity; in regulating and portioning water and solute flow and in storing and recycling nutrients. As an approximation of the capacity of ecosystems in each grid cell to maintain soil quality of we used a soil organic carbon content map (Jones et al., 2005).

Opportunities for recreation and tourism. Cultural ecosystem services are defined as the nonmaterial benefits obtained from ecosystems, among these the recreational pleasure that people derive from natural or managed ecosystems is defined as recreation service. Natural and semi natural ecosystems as well as cultural landscapes provide a source of recreation for humans. People enjoy forests, lakes or mountains for hiking, camping, hunting, fishing or bird watching or simply for their existence. The capacity of ecosystems in each grid cell to provide recreational services was mapped at 100 m resolution with the assumption that it is positively correlated to the degree of naturalness, presence of protected areas, presence of lakeshores and coastlines, and quality of bathing water (Paracchini et al., accepted).

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Table S1. Vertebrate's functional traits used to build a functional dendrogram of each taxonomic group and estimate the functional distinctiveness (FD) of each vertebrate species.

Type of traits	Traits	Variable type
Morphological	Bodymass (mammals, birds) Bodylength (amphibians, squamates)	Continuous
Diet	Mushrooms Mosses/Lichens Seeds/Nuts/Grains Fruits/Berries Vegitative Invertebrate Fish Small mammal Large mammal Herptile Bird/eggs Small bird Large Bird Vertebrate Bones Carrion Coprofagus	Categorical
Feeding behaviour	Opportunistic Hunting Browser Grazer	Categorical
Activity	Nocturnal Crepuscular Diurnal Arithmic	Categorical
Nesting location	Viviparous Elevated Tree/hole/fissure/in/the/bark Ground Rocks Building/Artificial Underground/water Cave/Fissures/Borrows Lodge Temporary/water Brooks/springs/small/rivers Puddles/ponds/pools/small/lakes Brackish/waters	Categorical

Figure S1. Ranking comparisons for the alternative biodiversity scenario (SP, SP scenario, EDFD, EDFD scenario, ED, ED scenario, FD, FD scenario). Upper panels correspond to the R^2 from the linear regression when comparing pairs of scenarios. The stars represent the significance level of the regression (*** = $p < 0.001$).

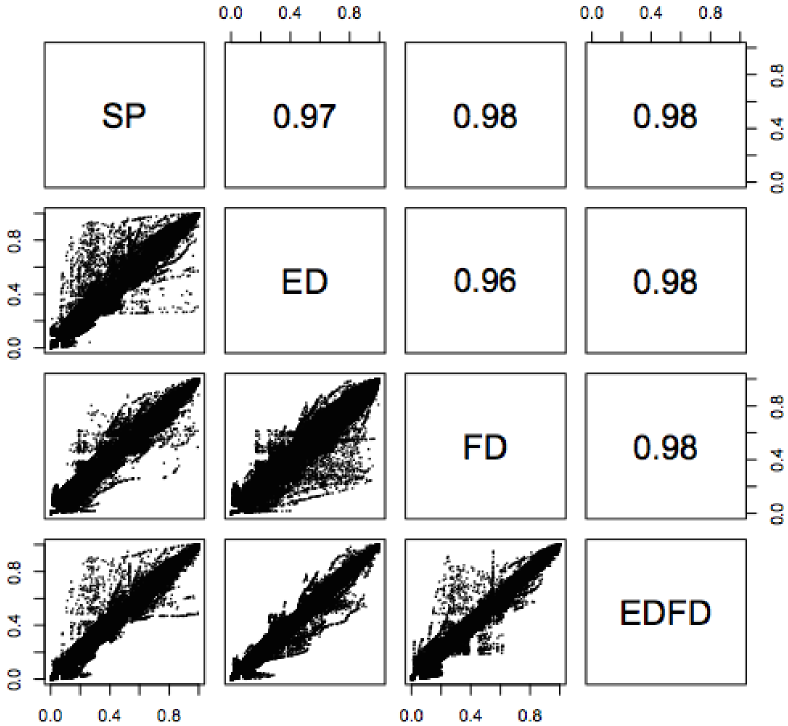


Figure S2. Feature's representation in each alternative conservation scenario. The first row corresponds to the representation of the most evolutionary distinct species in each scenario, second row correspond to the most functionally distinct species and third row correspond to the representation of the individual ES.

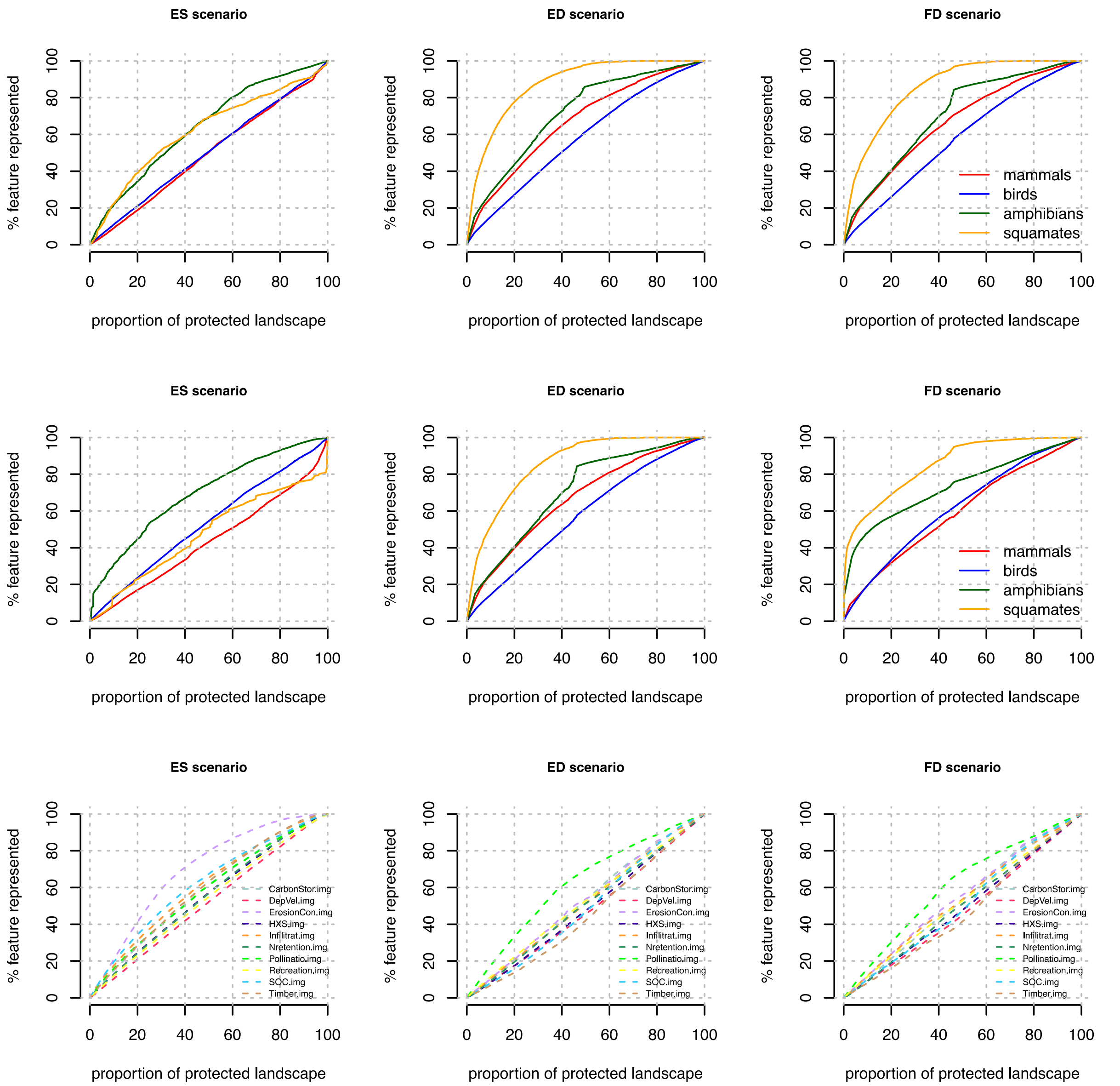


Table S2 Percentage of areas needed to reach 10%, 25%, 50% and 75% of representation of (A) ES and (B) biodiversity in both scenarios.

(A)	**All ES**		
	percent of EU27 needed		
	in the ES scenario	in the EDFD scenario	Difference
reach 10% of ES represented	6.33%	9.43%	3.1%
reach 25% of ES represented	16.62%	23.92%	7.3%
reach 50% of ES represented	36.32%	48.81%	12.49%
reach 75% of ES represented	61.71%	73.1%	11.39%

(B)	**All species**		
	percent of EU27 needed		
	in the EDFD scenario	in the ES scenario	Difference
reach 10% of species represented	1.53%	7.23%	5.7%
reach 25% of species represented	7.33%	18.62%	11.29%
reach 50% of species represented	25.72%	41.32%	15.6%
reach 75% of species represented	52.61%	68.61%	16%

Table S3. Mean observed values of representation of each taxonomic groups in (A) the EDFD scenario and (B) the ES scenario compared to their mean expected value of representation in a set of 100 random ranking.

(A) Null model for individual taxonomic groups in the ED FD scenario

	Mammals	Birds	Amphibians	Squamates
Observed				
mean	0.629***	0.599***	0.705***	0.862***
± sd	0.273	0.286	0.237	0.194
Expected				
mean	0.471	0.471	0.474	0.307
±sd	0.003	0.002	0.009	0.010

(B) Null model for individual taxonomic groups in the ES scenario

	Mammals	Birds	Amphibians	Squamates
Observed				
mean	0.486***	0.533***	0.676***	0.578***
± sd	0.278	0.286	0.262	0.256
Expected				
mean	0.471	0.471	0.474	0.307
±sd	0.003	0.002	0.009	0.010

Table S4. Mean observed values of representation of individual ES in (A) the ES scenario and (B) the EDFD scenario compared to their mean expected value of representation in a set of 100 random ranking. Blue values are for observed values (obs) superior to the expected value (exp).

(A) result null model individual ES in ES

	Climate regulation	Air quality regulation	Erosion control	Water provision	Water regulation	Water quality regulation	Pollination	Recreation	Soil quality regulation	Timber production
Obs mean	0.636***	0.513***	0.710***	0.546***	0.606***	0.563***	0.592***	0.547***	0.631***	0.621***
±sd	0.309	0.293	0.287	0.293	0.284	0.297	0.285	0.295	0.278	0.306
Exp mean	0.511	0.487	0.528	0.477	0.455	0.497	0.477	0.498	0.466	0.534
±sd	0.002	0.001	0.004	0.001	0.002	0.001	0.003	0.001	0.003	0.003

(B) result null model individual ES in EDFD

	Climate regulation	Air quality regulation	Erosion control	Water provision	Water regulation	Water quality regulation	Pollination	Recreation	Soil quality regulation	Timber production
Obs mean	0.525***	0.468	0.532***	0.489***	0.528***	0.502***	0.620***	0.518***	0.485***	0.459***
±sd	0.301	0.290	0.291	0.293	0.293	0.291	0.291	0.287	0.307	0.299
Exp mean	0.511	0.487	0.528	0.477	0.455	0.497	0.477	0.498	0.466	0.534
±sd	0.002	0.001	0.004	0.001	0.002	0.001	0.003	0.001	0.003	0.003

Table S5. Mean observed number of overlapping cells compared to the expected number of overlapping cells for selected fractions of protected landscape. *** p<0.001

Fraction of the landscape	1%	5%	10%	25%	50%	75%	100%
Observed	37***	345***	1013***	4244***	13849***	26411***	43014***
Expected	4	109	429	2689	10754	24185	43014

Figure S3. Comparison of the mean values of each ES across the landscape. The black crosses correspond to the mean value, while each black lines within the boxes are the medians.

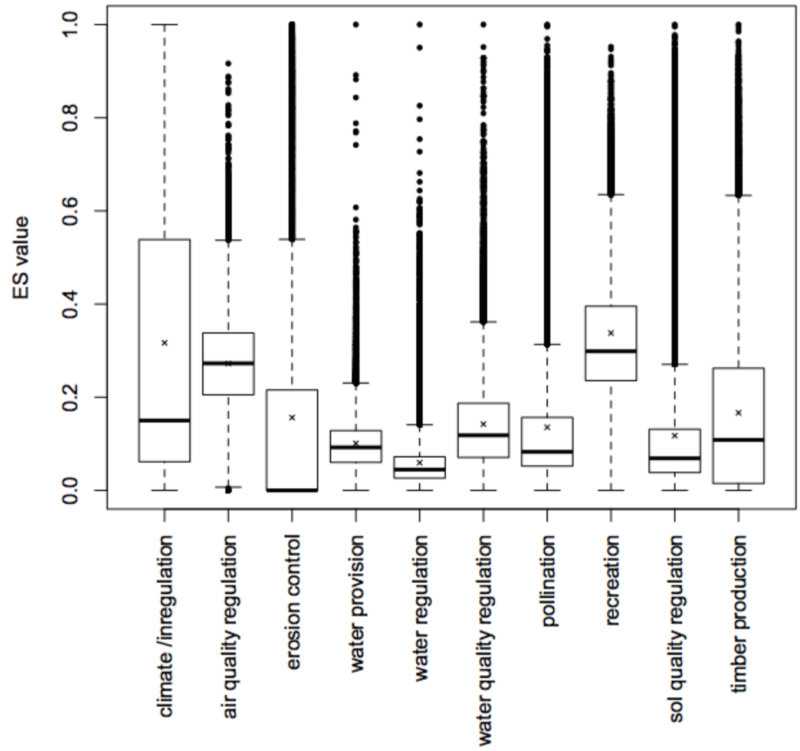
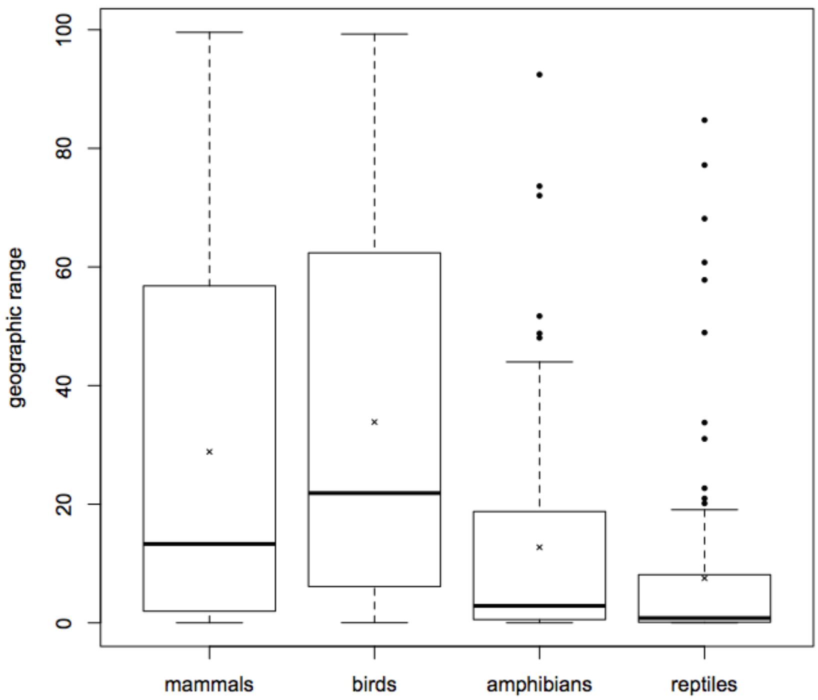


Figure S4 Comparison of the geographic range among taxonomic groups. By geographic range we mean the percentage of cells occupied by the species relative to the total number of cells of the study areas (EU27). The black crosses correspond to the mean value, while each black line within the boxes are the medians.



B. Supporting Information

1. Individual analysis following the CONNECT criteria

Table S1: Individual analysis of regulatory instruments following CONNECT criteria. Green backgrounds indicate a positive conclusion while red ones denote negative judgments.

	UTN	SRCE	PTCA
Effectiveness	No net environmental gain - By-pass strategies	Too recent	Effective in setting a broad international frame
Efficiency	Overall positive, even though qualified by limited effectiveness and associated high costs	Low degree of enforceability enables higher operational implementation at regional scale	High costs regarding current critics on low actual implementation (low enforceability)
Equity	Discrimination of small municipalities (costs) - Impede specific land allocations (i.e. constraints particular stakeholders)	High, thanks to a participative and iterative process for its design	Diminish uncontrolled development of mass tourism infrastructures - Toward more balance uses of sensitive areas
Legitimacy	Impartiality challenged - Scarcity of available information	Unchallenged to date	Unchallenged to date, except regarding the most appropriate scale of implementation
Consistency	Good current articulation - Attention for future adaptation	Good coherence with current dynamics of the policy mix - Good articulation among scales	Good articulation with regional and supra-regional dynamics (even one of its driver)
Complemen-tarity	High, mostly with land planning documents	Favouring synergies as one of its explicit objective (actual synergies will be assessed later)	High, with instruments at multiple scales
Absence of overlap and/or conflicts	Multiple overlaps within the administrative layer cake	Positive redundancy with protective perimeters at lower scale	Conflicts between broad objectives and local issues
Monitoring and control	Procedures of control exist but can be by-passed and real outcomes are not easy to access	Precise and adequate procedures are planned	Monitoring procedures exist but critics against superficial actual control

Table S2: Individual analysis of economic instruments following CONNECT criteria. Green backgrounds indicate a positive conclusion while red ones denote negative judgments.

	PNAL	PDR	PHAE2
Effectiveness	High, regarding actual wolf abundance - Enhanced cohabitation with stakeholders	Currently disputed, but should increase at long term through broad-based application of good practices	Low net environmental gain but high positive impact on maintain of extensive agriculture
Efficiency	Important budget for the protection of a single species	High costs regarding current critics on low current environmental gain	High costs for supporting mostly existing practises
Equity	Compensate for the additional constraints faced by farmers	Support the development of rural areas	Compensate for natural handicaps and existing premiums in more intensive agricultural contexts
Legitimacy	Instrument unchallenged as a tool of compromise - Many controversies remain on acceptation of wolf presence	Content unchallenged - Criticisms on insufficient articulation between local and regional scales	Unchallenged to date
Consistency	Consistency is not questioned	Good driver for transforming agricultural practises - Lack of local adaptation possibilities	Good adaptation of the measure over time in relation to the global Common Agricultural Policy dynamics
Complementarity	Many complementarities with instruments favouring extensive farming	Many complementarities between European premiums articulated at different scales	Many complementarities with instruments favouring alpine agriculture
Absence of overlap and/or conflicts	Supporting wolf presence induce additional constraints that conflict with many other instrument objectives	Good management of overlaps to limit negative effects - Numerous positive redundancies	No overlap or conflict detected
Monitoring and control	Precise and adequate procedures are followed	Precise and adequate procedures are followed	Auto-control by farmers on their practises - No strict and very precise procedures

Table S3: Individual analysis of voluntary instruments following CONNECT criteria. Green backgrounds indicate a positive conclusion while red ones denote negative judgments.

	IG	AeA	PAEN	ENS
Effectiveness	Environmental gain is not the initial objective but is actually supported	Support mountain agriculture but no environmental objective and very limited dimension of the project	Too recent (but seen as effective for limiting artificialisation)	Good ecological territorial coverage - Widespread instrument
Efficiency	Limited costs for an interesting comparative advantage	Very limited costs - Additional income source	High initial costs but perennial and cost-effective solution once established	High initial costs but perennial and cost-effective solution once established
Equity	Support product differentiation	Favour the diversification of activities (tourism) for farmers excluded before	Impede specific land allocations (i.e. constraints particular stakeholders)	Access to public is favoured even though it impedes alternative land allocations
Legitimacy	Unchallenged - Anchored in the cultural identity of the territory	Legitimacy anchored in the importance of pastoralism in the social and ecological mountain setting	Legitimacy linked to the collaborative process of their design at the scale of the "département"	Unchallenged
Consistency	Coherent with both the political setting (notably at UE scale) and the demand from society	Innovative project coherent with the diversification of tourism activities in the Alps and with frame management objectives	Consistency is not questioned	Good overall coherence - Questions related to the articulation between planning (region / département) and implementation (local)
Complemen-tarity	High, with diverse instruments at various scales	High, with diverse instruments at various scales	High, mostly with land planning documents	High, mostly with land planning documents and biodiversity conservation objectives
Absence of overlap and/or conflicts	No overlap or conflict detected - Positive redundancies detected	Positive redundancies supporting mountain farmers	Overlap with protective zoning for agricultural areas (ZAP) - Positive redundancy with ENS	Mostly positive redundancies
Monitoring and control	Precise and adequate procedures are followed - Required to maintain consumer's confidence	Good procedures for monitoring - Control not required due to the scale and specificities of the project	Overall satisfying even though environmental data is limited	Overall satisfying even though environmental data is limited

2. Individual analysis following a rebound effect analysis

Table S4: Individual analysis of regulatory instruments following a rebound effect analysis. Green backgrounds indicate a positive conclusion while red ones denote negative judgments. White backgrounds are for rebounds that were not particularly faced by the instruments and orange ones indicate variable impacts depending mostly on project charcateristics.

	UTN	SRCE	PTCA
Biodiversity Rebound I	Alteration of water cycle for artificial snow can impact downstream biodiversity if the design of the UTN procedure is not precise enough.	Increased pressure on non-prioritized areas→ decreased biodiversity	
Biodiversity Rebound II		Species with strong migration ability could be favoured over less mobile species	Focus on iconic species
Ecological Rebound	Negative impacts of artificialisation and tourist frequentation on ecological functioning --> impacts on water quantity and quality – habitat destruction - …	What impacts of increased connectivity? (e.g. changes in species communities and thus on ecosystem functions due to increased migration of certain species, including invasive species)	
Service Rebound	Trade-offs between all categories of ES strongly depend on project specificities	Trade-offs between all categories of ES strongly depend on project specificities	Positive for cultural ES
Environmental Rebound	Increase in visitor numbers → increase of greenhouse gas emissions, of CO_2-intensive energy consumption, of water pollution…		

Table S5: Individual analysis of economic instruments following a rebound effect analysis. Green backgrounds indicate a positive conclusion while red ones denote negative judgments. White backgrounds are for rebounds that were not particularly faced by the instruments and orange ones indicate variable impacts depending mostly on project charcateristics.

	PNAL		PDR		PHAE2	
Biodiversity Rebound I			Focus on mountain areas vs other rural areas in a budget-constrained context concentrates environmental-friendly measures		Concentrates measures on disadvantaged areas and on some specific parcels: spatial spill-over of impacts from more intense practices	
Biodiversity Rebound II	Impacts from different managements of mountain pastures (more intensively grazed areas → decreased biodiversity vs. areas abandoned and encroached → changes in Biodiversity) Overgrazing and trampling promote more generalist plant species over former ones		Favours a greater functional diversity (e.g. pollinators)	Depending on species habitat preferences, the maintain of open agricultural habitats can be positive or negative	Favours a greater functional diversity (e.g. pollinators)	Depending on species habitat preferences, the maintain of open agricultural habitats can be positive or negative
Ecological Rebound	Trophic cascade effects on ecosystems (positive regulation of food webs)		Positive effect on ecological functioning at mid- or long-term (but initial management can be challenging in previously intensively cultivated areas)		Positive effect on ecological functioning at mid- or long-term (but initial management can be challenging in previously intensively cultivated areas)	
Service Rebound	Negative for some cultural ES (recreation [protective herd dogs], aesthetic [land closure], hunting [competition]) and provisioning ES (food production)	Positive for some cultural ES (iconic species)	Positive for cultural and regulating ES	Negative for provisioning ES	Positive for cultural and regulating ES	Negative for food production (provisioning ES)
Environmental Rebound					If impacts on yields are important → imports of forage → Spatial environmental rebound and GES emissions.	

Table S6: Individual analysis of voluntary instruments following a rebound effect analysis. Green backgrounds indicate a positive conclusion while red ones denote negative judgments. White backgrounds are for rebounds that were not particularly faced by the instruments and orange ones indicate variable impacts depending mostly on project charcateristics.

	IG	AeA	PAEN	ENS
Biodiversity Rebound I	Inequality between rather similar territories can put higher pressure on non-certificated areas		Participates to maintaining high mountain pastures Increased urbanisation pressure on neighbouring lowland areas	Increased urbanization pressure on neighbouring areas
Biodiversity Rebound II	Explicit support for specific local species: benefits to genetic diversity		Prioritization of biodiversity aspects (certain species, certain habitats...)	Prioritization of biodiversity aspects (certain species, certain habitats...)
Ecological Rebound	Lack of coherence in supply chain can affect ecosystem functioning			Potential impacts from public access on sensitive natural areas
Service Rebound	Strong and auto-reinforcing synergy between cultural and provisioning ES Indirectly benefitting to regulating ES through demanding land management	Strong and auto-reinforcing synergy between cultural and provisioning ES	Expected to benefit to all types of ES	Synergy between regulating and cultural ES (educational value + iconic value) Potential trade-offs between regulating and cultural ES depending on sensitivity of the area
Environmental Rebound				

C. Detail of the supporting references consulted for the individual assessment of policy instruments

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IV. Appendices from Chapter IV

Presentation leaflet for the ICARE project

ICARE Cluses Arve et Montagnes

Information et Concertation sur l'Aménagement et les Ressources Environnementales

Favoriser la prise en compte des **ressources environnementales**, agricoles et forestières, dans les **projets d'aménagement** du territoire

Pour quoi ?

Par une **démarche scientifique de cartographie et d'analyse conjointe** des multiples ressources et fonctions des écosystèmes

Comment ?

1 projet / 4 questions

Où ?

Sur le territoire
De la communauté de communes Cluses Arve et Montagnes

Avec qui ?

L'ensemble des acteurs locaux intéressés, de manière **concertée**

Le territoire de projet

Cluses Arve & montagnes
Territoire de réussites

De sérieux atouts

- Des **espèces** et **paysages** patrimoniaux
- Une **agriculture** de montagne à valoriser
- Une attractivité **touristique**
- Des tissus **économiques** et **associatifs** dynamiques

Des points de vigilance forts

- Une **périurbanisation** croissante entrainant un **mitage** paysager, écologique et agricole
- Une **atmosphère** chroniquement polluée
- Une **planification territoriale** encore peu coordonnée

Un état des lieux des écosystèmes naturels et cultivés

Basé sur une expertise scientifique reconnue

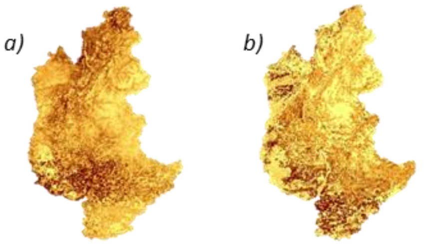
- Par la **cartographie** des différentes propriétés des **écosystèmes de montagne**
- En vue d'une analyse des différents **profils écologiques** caractéristiques du territoire

A transposer localement

- Pour **améliorer la connaissance** sur le fonctionnement écologique du territoire
- Pour **mettre en lumière** les ressources et les fonctions des écosystèmes
- Pour **dialoguer avec les acteurs du territoire sur leurs besoins et priorités**

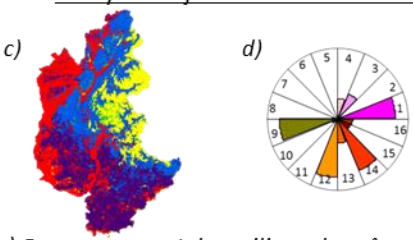
Exemples sur les Alpes (LECA – CNRS)

Cartographie individuelle des propriétés



a) Présence d'espèces animales protégées
b) Limitation de l'érosion des sols

Analyse conjointe sur le territoire



c) Regroupement des milieux de même fonctionnement écologique
d) Caractérisation d'un profil écologique

Une approche partenariale originale

A l'origine, un groupement d'acteurs constitué d'élus, de socio-professionnels et d'associations de protection de la nature et de l'environnement désireux **d'agir de manière innovante au service du territoire** afin de concilier aménagement du territoire et préservation des ressources environnementales.

Notre ambition : **donner un sens concret à la concertation en vous invitant à participer à notre groupe de travail.**

Rejoignez-nous !

Contact : FRAPNA Haute-Savoie 04 50 57 70 48

Projet soutenu par Fondation de France

FRAPNA

LECA Laboratoire d'Ecologie Alpine

Abstract

Abstract

Addressing trade-offs and synergies among ecosystem services and biodiversity
A multi-dimensional approach of their interactions
in the French Alps social-ecological system

In the context of global climate change and local land use changes, the future of the French Alps cultural landscapes, shaped through long-lasting and mutual interactions between human and their environment, appears uncertain. Simultaneously, the ecosystems constituting alpine landscapes host a rich biodiversity and provide the many natural resources and ecological functions that benefit to human societies. These resources and functions are conceptualised as "ecosystem services" and currently attract an increasing attention for the management and the conservation of environmental resources, along with biodiversity. Identifying the variables linked to their maintenance, in ecological, socio-cultural and political terms, is a necessary step of their sustainable management, and yet is still under-explored. My PhD project aimed at increasing the understanding of positive (synergies) and negative (trade-offs) interactions among ecosystem services and biodiversity through a multi-dimensional approach of the French Alps social-ecological system.

- In Chapter I, I present a **quantitative and spatially explicit biophysical assessment of ecosystem multifunctionality**. After a modelling step, we explored spatial patterns of trade-offs and synergies among ecosystem services and biodiversity using a series of statistical analyses of increasing complexity. Results were structured to provide insights for sound environmental governance at multiple scales. We identified various bundles of ecosystem services representative of the different conditions across the French Alps massif in terms of biogeography, management and landscape heterogeneity.
- This approach is complemented in Chapter II by a **qualitative representation of influence relationships among ecosystem services and biodiversity** that also accounts for additional ecological and social variables. We explicitly considered the multiple dimensions encompassed by the ecosystem service concept (their 'facets') and proposed an innovative conceptual framework to represent their influence networks. This framework was applied to analyse a consultative process that we carried out with stakeholders of regional expertise. This analysis highlighted their general perception of important influence relationships in the alpine social ecological system.
- In order to better understand social regulations linked to environmental governance, we test in Chapter III a methodology for **assessing the environmental effectiveness of policy instruments**. We concentrated on a restricted set of instruments regulating the interactions between biodiversity, agriculture and outdoor tourism. The consideration of multiple indicators assessing the performance and the fit with the socio-cultural and governance setting highlighted the complex articulation of instruments within the broader policy mix. Results were synthesised in a policy brief targeting regional decision-makers.
- Chapter IV is conceived as my **personal exploration of the conceptual and ethical issues** linked to research on ecosystem services. Following some general thinking on the relations between environmental sciences and society, I conducted a personal reflexive assessment of the research projects I contributed to.

To conclude, I propose a synthetic vision of the alpine social-ecological system and discuss the major issues revealed throughout the analyses.

Key words: bundles of ecosystem services, biodiversity, French Alps, biophysical analysis, influence networks, policy mix analysis, epistemic commitment.

Etude des compromis et synergies entre services écosystémiques et biodiversité
Une approche multidimensionnelle de leurs interactions
dans le socio-écosystème des Alpes françaises

Dans un contexte de changement climatique global et d'évolution locale de l'usage des terres, le devenir des paysages culturels des Alpes françaises, façonnés au cours des siècles par les interactions mutuelles entre sociétés et environnement, apparaît incertain. Dans le même temps, les écosystèmes qui les constituent abritent une biodiversité riche et sont à l'origine de nombreuses ressources naturelles et fonctions écologiques dont bénéficient les populations humaines. Ces ressources et fonctions sont conceptualisées sous le terme de « services écosystémiques » et font aujourd'hui l'objet d'une attention accrue dans la gestion et la protection des ressources environnementales, au même titre que la biodiversité. L'identification des facteurs liés à leur maintien, en termes écologiques, socio-culturels et politiques, est une étape nécessaire à leur gestion durable, bien qu'encore insuffisamment explorée. Mon projet de thèse visait à accroître la compréhension des interactions positives (synergies) et négatives (antagonismes) entre services écosystémiques et biodiversité via une approche multidimensionnelle du socio-écosystème des Alpes françaises.

- Le Chapitre I propose une **approche biophysique quantitative et spatialisée de la multifonctionnalité des écosystèmes**. Suite à une étape de modélisation, les patrons spatiaux de synergie et d'antagonisme entre services et biodiversité ont été explorés statistiquement et reliés à des enjeux de gouvernance actuels à différentes échelles. Ce travail a permis d'identifier les bouquets de services écosystémiques représentatifs des différentes conditions biogéographiques, de gestion et de d'hétérogénéité du paysage représentées dans le massif.
- Cette approche est complétée dans le Chapitre II par une **représentation qualitative des relations d'influence entre services écosystémiques et biodiversité**, ainsi que de leurs liens avec d'autres variables écologiques et sociales. Nous avons considéré explicitement les dimensions multiples englobées par le concept de service écosystémique (leurs 'facettes') et proposons un cadre conceptuel pour en cartographier les réseaux d'influence. Ce cadre a servi de base à l'analyse d'un processus consultatif que nous avons mené auprès d'acteurs du territoire. Les analyses ont mis en lumière leur perception globale des relations d'influence importantes au sein du socio-écosystème.
- Afin de mieux comprendre les régulations sociales appliquées à la gestion environnementale, nous testons dans le Chapitre III une méthodologie d'**analyse de l'efficacité environnementale d'instruments de gouvernance**. Notre analyse a privilégié un nombre restreint d'instruments qui encadrent actuellement les interactions entre agriculture, tourisme et biodiversité. L'utilisation d'un ensemble d'indicateurs de performance et d'adéquation avec le cadre socio-culturel et de gouvernance a souligné l'articulation complexe des instruments entre eux et a abouti à la production d'une synthèse pour les décideurs ('policy brief').
- Le Chapitre IV explore enfin certains **enjeux conceptuels et éthiques** de la recherche dans le domaine des services écosystémiques. Après une réflexion générale sur les relations entre science et société, je propose une évaluation réflexive et personnelle des projets de recherche auxquels j'ai contribué.

Pour conclure, je propose une vision transversale du socio-écosystème alpin mettant en lumière les enjeux majeurs identifiés par les différentes analyses.

Mots clés : bouquets de services écosystémiques, biodiversité, Alpes françaises, analyse biophysique, réseaux d'influence, analyse de gouvernance, engagement épistémique.

Abstract