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Tese orientada pelo Professor Doutor Henrique Miguel Pereira, especialmente elaborada para a obtenção do grau de doutor em Biologia (Especialidade de Biologia da Conservação)

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Resumo:

A alteração contínua dos ecossistemas Europeus por milénios de agricultura teve grandes impactos em vários níveis da biodiversidade e também do fornecimento de serviços de ecossistema. Em primeiro lugar, esta alteração contínua levou a enormes modificações da paisagem e a uma redução drástica da quantidade de áreas selvagens. Por outro lado, o desenvolvimento da agricultura levou também a uma mudança nos regimes de perturbações naturais. Em particular, a dinâmica natural de incêndios foi alterada, enquanto espécies de plantas agrícolas e espécies de animais domesticados substituíram progressivamente espécies de plantas e de animais selvagens. Para além disso, as alterações no uso do solo que foram impulsionadas pelas sociedades humanas levaram à perda e à fragmentação de habitat de algumas espécies, ao mesmo tempo que criaram novos habitats para outras espécies, influenciando tanto a sua distribuição como a sua abundância. Em paralelo, esta interacção secular entre os seres humanos e os ecossistemas teve uma forte influência no património cultural Europeu.

No entanto, os aumentos recentes de concorrência no mercado agrícola levaram ao êxodo rural e ao abandono dos solos menos produtivos do continente. De forma a aumentar a sua competitividade e a manter alguma actividade nas suas terras, os proprietários e gestores dos terrenos recorrem normalmente a uma de três estratégias de gestão: intensificação, extensificação e arborização. Esta tese investiga uma quarta opção, passiva, de gestão dos terrenos: o retorno ao estado selvagem, ou *rewilding*.

Rewilding é uma forma de restauração passiva, cujo objectivo final é o de restaurar ecossistemas para que estes se tornem auto-sustentáveis. Ao contrário de outras práticas de conservação que são centradas na restauração de determinados habitats ou de serviços de ecossistema específicos, o *rewilding* foca-se na restauração dos processos e, ainda, na restauração da resiliência dos ecossistemas.

O abandono agrícola e as suas consequências na paisagem e biodiversidade geram

preocupações tanto a nível da comunidade científica como na população em geral. Como resultado destas preocupações, a opção de *rewilding* raramente é considerada como uma ferramenta para a gestão do espaço rural. Esta tese de doutoramento aborda diversos aspectos do *rewilding*, a fim de o integrar na agenda de conservação da Europa para as próximas décadas.

A primeira parte do estudo investiga o potencial para o *rewilding* no contexto do abandono agrícola na Europa. Em primeiro lugar foram analisados os paradigmas da agricultura na Europa, e qual a probabilidade do abandono agrícola continuar nas próximas décadas. Usando os resultados do modelo CLUE, estimamos que a quantidade de solo que poderá ser convertida de solo agrícola para natural, até 2030, possa vir a ser tão elevada quanto 29.7 milhões de hectares. Em particular, os locais principais de abandono das terras e *rewilding* foram identificados maioritariamente em regiões montanhosas e marginais, encontrando-se não só nos Alpes, nos Apeninos e no norte de Portugal, mas também no noroeste da França e na Europa Central. Este trabalho discute também os desafios associados à aplicação do *rewilding* e investiga as consequências do mesmo para o fornecimento de serviços de ecossistema, bem como as consequências que o rewilding poderá ter para a biodiversidade. Relativamente a esta última, foram identificadas mais de 100 espécies que potencialmente beneficiariam com o *rewilding*, algumas das quais mostrando já tendências positivas no que diz respeito à sua distribuição e à sua abundância.

O segundo estudo foca-se na interacção entre as sociedades humanas e a paisagem em que estas sociedades habitam. Embora os motores do abandono das terras e as mudanças na paisagem que resultam deste abandono sejam conhecidos, é ainda rara a utilização de dinâmicas socioecológicas integradas de alteração no uso do solo. Neste sentido, usamos uma abordagem teórica para investigar a interacção entre uma comunidade composta por nómadas ou por sedentários e a quantidade de área de floresta existente no território. Este modelo foi baseado no caso da migração sazonal entre aldeias de verão e aldeias de inverno (brandas e inverneiras), observada na freguesia de Castro Laboreiro, no norte de Portugal. O modelo socioecológico ilustra a dinâmica interdependente da população sedentária e da cobertura florestal, tanto nas aldeias de verão assim como nas aldeias de inverno. Para além disso, o modelo permite ainda identificar a existência de pontos de viragem no sistema, os quais não poderiam ter sido detectados através de um modelo apenas sociológico ou de um modelo apenas florestal.

A parte seguinte do estudo investiga como o *rewilding* poderia ser utilizado com o fim de superar os filtros ecológicos para a sucessão secundária resultante do legado de cultivo da terra. Em particular, a pesquisa apresenta como um *rewilding* assistido poderia restaurar os regimes de perturbações naturais que tinham sido substituídos por actividades humanas que foram entretanto abandonadas, de forma a manter uma paisagem heterogénea mas no entanto selvagem. Esta restauração poderia ser feita através do uso de incêndios controlados e através do reforço ou da reintrodução de populações de herbívoros selvagens, de forma a despoletar os regimes de perturbações naturais e ainda de modo a manter um mosaico de habitats. O estudo investiga ainda como as perturbações moldaram as paisagens Europeias e a resultante comunidade de espécies que habitam estas paisagens.

A quarta parte desta tese de doutoramento avalia a contribuição do *rewilding*, e do aumento consequente de áreas selvagens, para o fornecimento de serviços de ecossistema a nível da Europa. Esta avaliação foi realizada em primeiro lugar através de uma análise qualitativa, realizada à escala europeia, a qual sobrepôs o fornecimento dos serviços de ecossistema e o índice de qualidade de áreas selvagens. Em seguida foi realizada uma análise quantitativa, à escala ibérica, a qual comparou o fornecimento de vários indicadores de serviços dos ecossistemas em terras agrícolas, áreas selvagens de elevada qualidade e solos que são actualmente cultivados, mas cujas projecções indicam que estarão abandonados em 2030. Os resultados deste estudo sugerem um aumento no fornecimento de vários indicadores de serviços de ecossistema. Em particular, o fornecimento de serviços de regulação (por exemplo, regulação da água e regulação do clima) e de serviços de recreação seria melhorado através de um aumento das áreas selvagens, resultante do abandono agrícola e de *rewilding*.

Finalmente, a última parte da pesquisa examina o *rewilding* na interface entre a ciência e a política na União Europeia, avaliando-o em paralelo com as Redes de Áreas Protegidas, com a Rede Natura 2000, com a implementação de regimes agro-ambientais e com o recente interesse político na conservação de zonas selvagens. Este estudo investiga ainda a contribuição potencial do *rewilding*, e do aumento consequente de zonas selvagens, para os objectivos de conservação estabelecidos para 2020 tanto a nível global como a nível da União Europeia. Como contribuição final desta pesquisa deixamos uma lista de recomendações orientadas para as políticas, a fim de implementar eficazmente o rewilding como uma ferramenta de gestão das terras com o objectivo da restauração.

Palavras-chave: retorno ao estado selvagem (*rewilding*), ecologia da restauração, áreas selvagens (*wilderness*), abandono das terras, dinâmica socioecológica, interface ciência-política, biodiversidade, serviços de ecossistema.

Abstract

The continuous alteration of the European ecosystems by millennia of agriculture had major impacts, on several dimensions of biodiversity and the supply of ecosystem services, at various scales. First, it led to tremendous modifications of the landscape, and a drastic reduction of the amount of wilderness. The development of agriculture also led to a change in the regimes of natural disturbance. In particular, the natural fire dynamics were altered, while domesticated plant and animal species progressively replaced wild species. Furthermore, the land-use changes driven by societies caused habitat loss and fragmentation for some species while creating new habitats for others, influencing both distributions and abundances. In parallel, this age-old interaction between humans and ecosystems strongly influenced the European cultural heritage.

Nonetheless, recent increases in the agricultural market competition have led to rural depopulation and land abandonment on the least productive soils of the continent. To increase their competiveness, and maintain an activity on their land, landowners and managers typically resort to one of three active management strategies: intensification, extensification, and afforestation. This thesis investigates a fourth, passive, land management option: rewilding.

Rewilding is a form of passive restoration with the ultimate goal to restore self-sustaining ecosystems. Unlike conservation practices centered on the restoration of given habitats or ecosystem services, rewilding is focusing on the restoration of processes, and the restoration of the ecosystem's resilience.

Farmland abandonment and its consequences on the landscape and for biodiversity are generating concerns in both the scientific community and the public. As a result, rewilding is rarely considered a tool for the management of the land. This thesis addresses several aspects of rewilding in order to integrate it in the conservation agenda of Europe for the decades to come.

The first part of the research investigates the potential for rewilding in the context of

farmland abandonment in Europe. This was done by first analyzing the paradigms of European agriculture and how likely farmland abandonment is to continue in the decades to come. Using the output of the CLUE model, we estimate that the amount of land that could be converted from agricultural to natural by 2030 could be as large as 29.7 Mha. In particular, the hotspots of land abandonment and rewilding were identified mainly in mountainous and marginal regions, in the Alps, the Apennines, Northern Portugal, but also in Northwestern France, and Central Europe. The research also discusses the challenges of applying rewilding and investigates the consequences of rewilding for the supply of ecosystem services and for biodiversity. Regarding the latter, over 100 species potentially benefiting from rewilding were identified, some of which already showing positive trends in their distribution and abundance.

The second study addresses the interaction between societies and the landscape in which they occur. As a matter of fact, though the drivers of land abandonment and the resulting changes in the landscape are understood, coupled socio-ecological dynamics of land-use change are still rarely used. A theoretical approach was used to investigate the interaction between a community composed of nomads and sedentary, and the amount of forested area in the landscape. This model was based on the case of seasonal migrations between summer and winter villages (brandas and inverneiras), observed in the parish of Castro Laboreiro in Northern Portugal. The socio-ecological model illustrates the coupled dynamics of the sedentary population and the forest cover in both the brandas and the inverneiras. Furthermore, the model allows us to identify the existence of tipping points in the system which could not have been detected using a sociological model, or a forest model alone.

The next part of the study researches how rewilding could be assisted in order to overcome the ecological filters to secondary succession resulting from the cultivation legacy of the land. In particular, the research presents how assisted rewilding could restore the regimes of natural disturbances that were replaced by human activities, which are now abandoned, in order to maintain a heterogeneous, yet wild, landscape. Such restoration could be done via the use of prescribed burning and the reinforcement or reintroductions of populations of wild herbivores, in order to trigger the regimes of disturbance and maintain a mosaic of habitats. In addition, the study investigates how disturbances shaped the European landscapes and the resulting community of species that inhabit them.

The fourth part of this PhD research evaluates the contribution of rewilded land, and the resulting increase in wilderness, to the supply of Ecosystem Services in Europe. Such assessment was first done with a qualitative analysis, at the European scale, which overlaid the supply of ecosystem services and the wilderness quality index. A quantitative analysis, at the Iberian scale, was then performed by comparing the supply of various indicators of ecosystem services in agricultural land, high quality wilderness, and land currently cultivated but projected to be abandoned by 2030. The results do suggest an increase in the supply of several indicators of ecosystem services. In particular, the supply of regulating (e.g. water and climate regulation) and recreational services would be improved by an increase in the wild areas resulting from agricultural land abandonment and rewilding.

Finally, the last part of the research examines rewilding at the science-policy interface in the European Union, by assessing it in parallel to the Nationally Designated Protected Areas, the Natura 2000 Network, the implementation of Agri-Environmental Schemes, and the recent policy interest in wilderness conservation. The study also investigates the potential contribution of rewilding and the resulting increase in wild lands, to the global and EU conservation targets established for 2020. A final contribution of this research is a list of policy-oriented recommendations in order to effectively implement rewilding as land management tool for restoration.

Thus, acknowledging the trends and projections for a new availability of land in the decades to come, the work presented in this thesis will serve to better define the framework of rewilding in Europe, and particularly its application to efficiently restore the abandoned agricultural lands, for the benefit of both biodiversity and human well-being.

Keywords: rewilding, restoration ecology, wilderness, farmland abandonment, socio-ecological dynamics, science-policy interface, biodiversity, ecosystem services

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Contents

1. Introduction

"All we have to decide is what to do with the time that is given us".
J R R Tolkien, 1954.

The first and second sections of the introduction are based on Pereira H.M., **Navarro, L.N.** and Martins I.S. (2012). *Global Biodiversity Change: The Bad, the Good, and the Unknown*, Annual Review of Environment and Resources, 37(1): 25-50 (see Appendix A)

The third section is partly based on Beilin, R., Lindborg, R., Stenseke, M., Pereira, H.M., Llausàs, A., Slätmo, E., Cerqueira, Y., **Navarro, L. M.**, Rodrigues, P., Reichelt, N., et al. (2014). *Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from Scandinavia, Iberia and Oceania.* Land Use Policy 36: 60–72. (see Appendix B)

Introduction

1.1. Habitat change and biodiversity alterations

Investigating the relationship between societies and their environment is a first important step to understand the status and trends of landscapes and the biodiversity that they sustain. Humans have interacted with their environment for millions of years, since the first use of fire. In Europe, the first evidence of a regular and controlled use of fire by hominins dates back to 400-300 000 BP (Roebroeks & Villa, 2011), though the use of fire for land and species management only started some tens of thousands of years ago (Bowman *et al.* , 2009). Aside from fishing and hunting, both having tremendous impacts on biodiversity (Pereira *et al.* , 2012), the next major source of anthropogenic impact on the European landscapes was the development of agriculture between 11-9000 BP and 8-6000 BP (Ellis *et al.* , 2013; Pinhasi *et al.* , 2005; Ruddiman, 2013). Again, fire was typically used to clear forests before establishing cultivated fields or pastures (Bowman *et al.* , 2009).

By 2300 BP, most of the European land was covered by less than half of its potential in forested areas (Kaplan *et al.* , 2009). The continuous alteration of ecosystems by human activities had major impacts, on several dimensions of biodiversity and at various scales (Ellis *et al.* , 2013; Pereira *et al.* , 2012). The increase of agricultural areas also meant an increase in the appropriation of the Net Primary Productivity by humans (Erb *et al.* , 2009; Krausmann *et al.* , 2013). Additionally, agriculture and landscape alteration led to a change in the fire regime (see chapter 4 and references therein). Finally, the domestication of plant and animal species led to a "replacement" of wild species (e.g. large herbivores, see chapter 4 and references therein).

Today, Europe is the continent with the least cover of wilderness (Kaplan *et al.* , 2009; Mittermeier *et al.* , 2003), while the Mediterranean Basin, where less than

5% of the primary vegetation cover remains, is considered a hotspot of conservation priority (Myers *et al.* , 2000).

According to the assessment of the Red List of the IUCN, 34 species occurring in Europe went extinct, or extinct in the wild (IUCN, 2011). Additionally, 76 species of birds, amphibians and mammals native to the continent are currently threatened (i.e. either Critically endangered, Endangered, or Vulnerable). Nonetheless, when compared to other regions of the world, Europe has on average a low proportion of threatened species (less or equal to 4%), with the exception of the Mediterranean Basin (Fig. 1.1A). Yet, 242 of the total number of species of birds, amphibians and mammals assessed by the IUCN and native to Europe are threatened by habitat loss (following the same selection criteria as in Pereira *et al.* , 2012). As a matter of fact, habitat change and degradation have been identified as a major cause of biodiversity alterations, and affects globally more than 80% of the species assessed by the IUCN (IUCN, 2011; Pereira *et al.* , 2012). Habitat change can be characterized by the conversion of natural habitat to human-dominated habitat, by the intensification of human activities on the land, or by the recovery of land abandoned by human activities (Pereira *et al.* , 2012). When looking at the proportion of species impacted by habitat loss, Europe, with the exception of the Northern parts of the continent, presents average values higher than 20% (Fig. 1.1B), which places the continent among those globally hosting more species at risk, if focusing on this single driver of change (see Fig.7a in Appendix A).

1.2. A window for optimism?

Positive messages for biodiversity conservation can arise from the observed success of conservation programs (Pereira *et al.* , 2012). This can be exemplified by the effectiveness of the Bird Directive, an EU conservation policy (see chapter 6), measured in the positive trends shown by some bird populations between 1970-1990 and 1990-2000 (Donald *et al.* , 2007).

Another successful example of conservation is illustrated by the programs of captive-breeding and reintroductions of European bison (*Bison bonasus*), which went from being nearly extinct in the 1950s to being estimated at 2 759 individuals in 2011, 40% of which in EU28 countries (Deinet *et al.* , 2013; Hoffmann *et al.* , 2011). Moreover, following the ban on the use of wildlife poisoning and the establishment

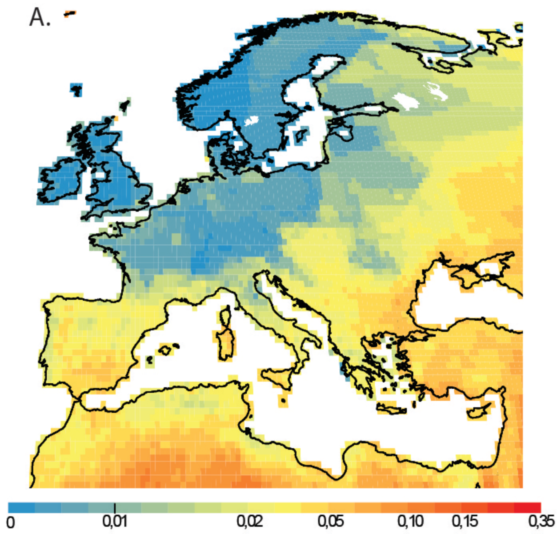

Figure 1.1.: European distribution of threatened species and species affected by habitat loss. Calculations are based on grid cells of 0.48º x 0.48º. The scale was intentionally left as in the original version of the figure as to be able to compare the European values to the rest of the globe. **A.** Proportion of threatened species (number of threatened species divided by the number of species in each cell). **B.** Proportion of species affected by habitat loss (residential and commercial development, agriculture and aquaculture, energy production and mining, transportation and service corridors, and natural system modifications). Adapted from IUCN (2011); Pereira *et al.* (2012).

of “Vulture restaurants” (i.e. feeding stations), the griffon vulture (*Gyps fulvus*) went from less than 5 000 breeding pairs in the mid 1980s to over 25 000 breeding pairs in 2012 (Deinet *et al.* , 2013; Cortés-Avizanda *et al.* , in press).

Another way to think positively about conservation is to observe the reversion of the effect of a driver of change (Pereira *et al.* , 2012), such as a decrease or a complete withdrawal of human pressure on the land, and their potential to reverse the impact of habitat loss and alteration. This phenomenon can be triggered by active conservation and restoration measures (Pereira *et al.* , subm, and see chapter 6).

Numerous cases of natural regeneration on abandoned land have been documented for the past decades, across the globe and in most biomes Pereira *et al.* (subm). Natural secondary successions are particularly successful if agricultural practices involved a short-term appropriation of the land (Bowen *et al.* , 2007; Hobbs & Cramer, 2007), yet most ecosystems can typically recover from human activities although the time needed for passive restoration can greatly vary (Aide & Grau, 2004; Rey Benayas *et al.* , 2007). Several regions of the globe are currently showing signs of “recovery” from abandoned land-uses, including in Europe (Ellis *et al.* , 2013; MA, 2005; Verburg & Overmars, 2009 and see chapter 2).

In particular, rewilding is a form of passive restoration on abandoned land with the ultimate goal to restore self-sustaining ecosystems (see chapter 2). Unlike conservation practices centered on the restoration of given habitats or ecosystem services, rewilding is focusing on the restoration of processes (Byers *et al.* , 2006; Pereira *et al.* , subm; Sandom *et al.* , 2013a), and the restoration of the ecosystem’s resilience. If the land is not too degraded, once abandonment occurred, natural regeneration can passively restore the systems, with human intervention only preventing the appropriation of the land for new human activities (Clewell & McDonald, 2009). Assisted passive restoration can accelerate rewilding when ecological filters would hamper it (Shono *et al.* , 2007, and see chapter 2 and chapter 4).

In order to make room for rewilding in the restoration policy agenda, we need to understand what drives land abandonment, as to predict where it is likely to happen in the next decades, and where rewilding could be a successful and efficient land management for conservation.

1.3. Trends and drivers of land abandonment

Land abandonment is driven by rural depopulation, which responds to several global and local factors. The abandonment of cultivated land has for example been observed in regions of the globe where shifts towards industry and services based economies are occurring (Aide & Grau, 2004). As a matter of fact, land abandonment has been positively correlated with the GDP (Rey Benayas *et al.* , 2007).

Although the human population considerably increased in the past century, the resulting higher demand for food has not caused a linear increase in agricultural areas (Ellis *et al.* , 2013), suggesting that food production and conservation could be compatible (Ellis, 2013; Phalan *et al.* , 2011; Tilman *et al.* , 2011). This phenomenon can be explained by the fact that high agricultural productivity and technological improvements allow for more food to be produced on less land (Kaplan *et al.* , 2009; Tilman *et al.* , 2011), though nutrient efficiency should be researched to limit the environmental impacts of intensification. Additionally, rather than food production, addressing the inequalities in food security and food sovereignty could limit hunger and poverty efficiently (Fischer *et al.* , 2014; Tscharntke *et al.* , 2012), without substantially increasing the global cultivated areas.

As a result, not only did the agricultural area not increase linearly with the human population, but it also decreased in some regions of the world (Ellis *et al.* , 2013; Ramankutty *et al.* , 2002; MA, 2005). This can be illustrated by the fact that, though during the 20th century, globally 600 million ha of land were converted to agriculture, 222 million ha of land previously cultivated were abandoned during the same period of time (Ramankutty *et al.* , 2002). In particular, we estimated that about 15% of the European agricultural area decreased between 1970 and 2010 (based on data from PBL, 2012, in Pereira *et al.* , subm).

The drivers of abandonment and their relative effect depend on the social and historical context of each area, and can act at the international, national and/or local scale (Beilin *et al.* , 2014). Typically, in Europe, the land being abandoned is less productive and under physical constraints, which makes it difficult to mechanize agriculture (MacDonald *et al.* , 2000; Rey Benayas *et al.* , 2007; Strijker, 2005). When combined with past strategies of the EU Common Agricultural Policy, which promoted intensification (Strijker, 2005), and pressures from global markets (Beilin *et al.* , 2014), remote and less productive soils became less and less competitive, and more prone to abandonment. In some cases, the abandonment was increased by

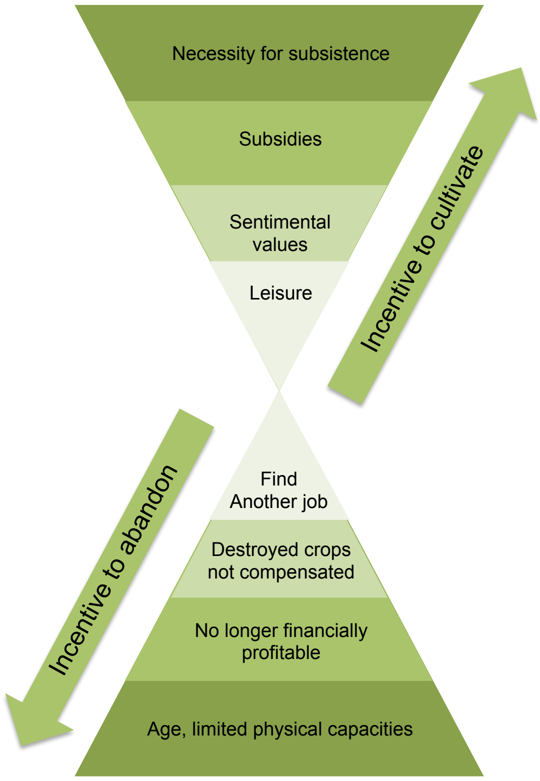

Figure 1.2.: Ranking of the different motives behind the abandonment and/or continuation of farming activities in the Castro Laboreiro Parish, from least important (inner) to most important (outer). The figure is based on data gathered during interviews performed between 2009 and 2010 (Cerqueira, 2014).

outmigration to other countries in the search of a better life situation (Beilin *et al.* , 2014; Pereira *et al.* , 2005). The progressive rural depopulation of those areas also creates a self-feeding circle of social decline, where basic amenities (e.g. schools, hospitals) and job opportunities become scarce, thus leading to more depopulation (Pereira *et al.* , 2005; Figueiredo & Pereira, 2011). Those drivers can be counteracted by various frictions to land abandonment, such as the development of tourism and leisure, cultural values and a sense of identity, and national rural development initiatives and subsidies (see Table 2 in Appendix B).

Interviews conducted within the population of the Peneda Gerês mountain range, in Northern Portugal, revealed that on the one hand the low profit for the cultivation of the land by an aging population are driving abandonment, while on the other hand, the necessity for subsistence and subsidies are major incentives for maintaining agriculture (Fig. 1.2). The social cohesion in an area is also a determining factor in the rate of land abandonment. As a matter of fact, within a tight community, people tend to mimic each other's behaviors (Goldstein *et al.* , 2008), which can further enforce the circle of social decline in remote areas (see chapter 3).

If the trends of rural depopulation and land abandonment continue as observed, or increase, there is a non-negligible potential for ecosystem restoration and biodiversity conservation (Ellis, 2013), and rewilding should be considered as a valid option. Nonetheless, in Europe, the abandonment of agricultural land and rewilding tend to be perceived negatively, by the public, and by the scientific community (see chapter 2).

1.4. Perceptions of land abandonment and rewilding

In a recent meta-analysis, (Queiroz *et al.* , 2014; Antrop, 2005) identified an important regional variation in the observed consequences of farmland abandonment for biodiversity. For example, in North America, the focus of the studies was on the restoration of ecosystems "post-abandonment", while in Europe, where most of the studies had been conducted, the focus was clearly oriented towards the restoration of ecosystems to a status similar to the "pre-abandonment" conditions. Consequently, the documented responses of biodiversity to land abandonment were mainly negative in Europe, unlike in North America.

These results comfort the idea that identifying the end goal of conservation, i.e. "what do we want to preserve and/or restore, and where?" is fundamental in a context of farmland abandonment and rewilding (e.g. Carvalho-Ribeiro *et al.* , 2013). Society's inclination towards given species or habitats depends on (subjective) baseline, which retrospectively dictate what should be preserved (Queiroz *et al.* , 2014; Antrop, 2005). Yet, those baselines are multiple (e.g. functional, historical, aesthetical, cultural) and can shift from generations to generations (Papworth *et al.* , 2009; Vera, 2009).

The fact that in Europe, most landscapes have not been "wild" for millennia (Ellis *et al.* , 2013; Kaplan *et al.* , 2009) can explain why most conservation is focused on maintaining the traditional agricultural landscapes (Fig. 1.3.A) which are now threatened by land abandonment (Queiroz *et al.* , 2014). At the opposite, the remaining wildlife is often perceived as a source of conflict (e.g. Kaczensky *et al.* , 2004; Wilson, 2004; Treves & Bruskotter, 2014 and see chapter 2), and/or a threat,

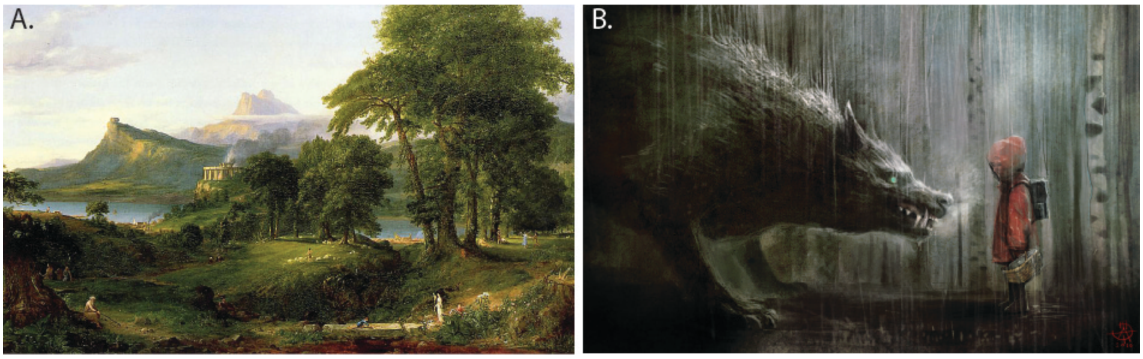

Figure 1.3.: Two opposing concepts of nature and wildlife. A. *The Arcadian or Pastoral state*, by Thomas Cole (1834). B. Illustration of *the Little Red Riding Hood* of Charles Perrault (1697), by Manuel Moura.

as portrayed in folklore and tales (Wilson, 2004; Boitani, 2000 and see Fig. 1.3.B). We are now even creating double standards in conservation biology, where developed countries preserve a human-influenced landscape, while developing countries are encouraged to limit their impact on the land and to preserve wildlife, regardless of conflicts (Meijaard & Sheil, 2011).

Yet, wild lands (either current wilderness or future rewilded areas) must become part of the conservation agenda, in Europe as in the rest of the globe. Wilderness areas provide a wide range of ecosystem services (chapter 5), which supply could be enhanced by successful rewilding management (see chapter 4). Rewilding can also benefit several species (see chapter 2), which became segregated to the few remaining wilderness areas (or adapted to human pressure).

Wilderness recently gained more importance in the EU conservation policy agenda (see chapter 6). This trend, added to the observed and expected land abandonment, could allow rewilding to gain a momentum in land management and restoration ecology.

1.5. Objectives and outline of the thesis

The objective of this thesis is to present rewilding as an opportunity to address the issue of land abandonment in Europe, and to contribute to global restoration targets. In particular, this thesis aims at addressing the following issues:

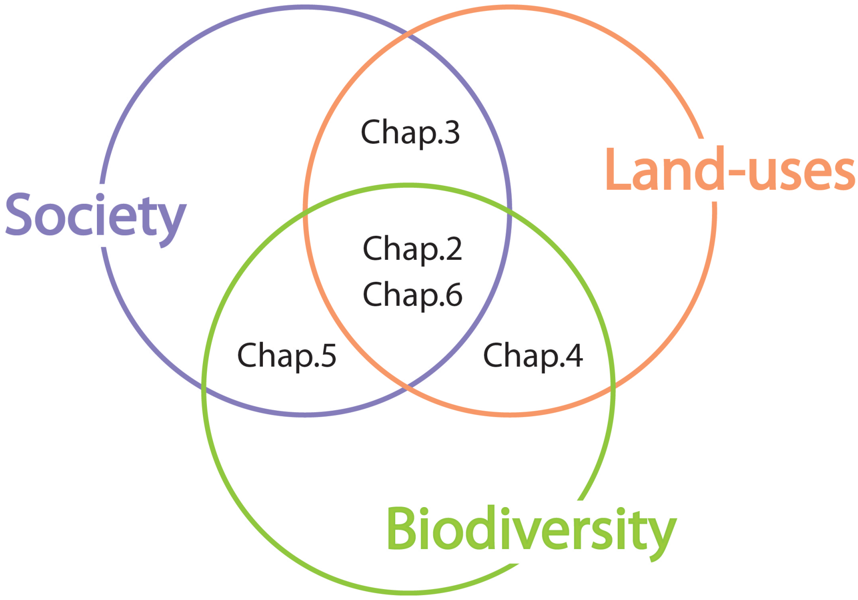

Figure 1.4.: Organisation of the chapters within the thesis

1. What is the current and projected extent of farmland abandonment in Europe?
2. How do the socio-economic aspects of abandonment influence the land cover, and what is the reciprocity of such interaction?
3. Can "assisted rewilding" overcome ecological filters to passive restoration?
4. Should land abandonment and rewilding be considered as a threat or as an opportunity for biodiversity?
5. Can human well-being be improved by rewilding and the resulting increase of wild areas in Europe?
6. Are the EU conservation policies adapted to farmland abandonment, and how can they integrate rewilding as a land management option?

The research presented in this thesis is situated at the interface between societies, land-uses, and biodiversity (Fig. 1.4). It is organised around five chapters, which topics relate to the interaction between society and landscapes (chapter 3), between society and biodiversity (chapter 5), between landscapes and biodiversity (chapter 4), or at the interface between the three (chapter 2 and chapter 6).

In chapter 2, we investigate the potential for rewilding in a context of farmland abandonment in Europe by addressing four main topics (some of which being further discussed in subsequent chapters of the thesis). The chapter first identifies which are the paradigms of European agriculture and how likely farmland abandonment is to continue in the decades to come. The consequences of land abandonment for biodiversity and the supply of ecosystem services are then studied. Finally, chapter 1 discusses the challenges of applying rewilding and the options to overcome them. The work presented in this chapter was published as: Navarro, Laetitia M. and Pereira, Henrique M. (2012) *Rewilding Abandoned Landscapes in Europe*, Ecosystems 15: 900-912.

In chapter 3, we present a socio-ecological model of sedentarization and its consequences on the forest cover. This model stems from the unique system of seasonal migrations between summer villages (*brandas*) and winter villages (*inverneiras*) historically observed in parishes of the Peneda Gerês National Park. The model investigates the coupled dynamic between the social system (i.e. balance between sedentary and nomads, drivers of sedentarization), and the ecological model (i.e. forest dynamic). The work presented in this chapter is currently a manuscript, to be submitted to Ecological Modeling as: Navarro L.M., Figueiredo J., Pereira H.M. (in prep). *A socio-ecological model of sedentarization: the case of brandas and inverneiras in Northern Portugal.*

In chapter 4, we investigate disturbances, which can be either anthropogenic (e.g. agriculture) or natural (e.g. fire, ecosystem engineers), and how they shaped the European landscapes and the resulting community of species that inhabit them. In a context of farmland abandonment and altered regimes of natural disturbances, the chapter presents how rewilding, in an assisted form when needed, can contribute to the maintenance of threatened habitats. The work presented in this section is the chapter of a book: Navarro L.M., Proença V., Kaplan J.O., Pereira H.M. (in press). *Maintaining disturbance-dependent habitats.* In: Rewilding European Landscapes, Pereira H.M. and Navarro L.N. (eds). Springer.

In chapter 5, we address the contribution of rewilded land and the resulting increase in wilderness to the supply of Ecosystem Services in Europe. Such assessment is done with both a qualitative analysis (at the European scale) and a quantitative analysis (at the Iberian scale), by comparing the supply of various indicators of ecosystem services in agricultural land, high quality wilderness, and land currently cultivated but projected to be abandoned by 2030. The work presented in this section is the chapter of a book: Cerqueira Y., Navarro L.M., Maes J., Marta-Pedroso C., Honrado J., Pereira H.M. (in press). *Ecosystem Services: the opportunities of rewilding in Europe.* In: Rewilding European Landscapes, Pereira H.M. and Navarro L.N. (eds). Springer.

In chapter 6, we examine rewilding at the science-policy interface in the European Union. In particular, the potential contribution of rewilding to EU biodiversity strategies is addressed, in parallel with the Nationally Designated Protected Areas, the Natura 2000 Network, the implementation of Agri-Environmental Schemes, and the recent momentum gained by wilderness. The work presented in this section is the chapter of a book: Navarro L.M. and Pereira H.M. (in press). *Towards a European policy for Rewilding.* In: Rewilding European Landscapes, Pereira H.M. and Navarro L.N. (eds). Springer.

Finally, the last chapter synthesises the main results of the previous chapters and addresses future perspectives for rewilding research in order to propose an adapted framework for rewilding as a restoration option on European landscapes.

2. Rewilding Abandoned Landscapes in Europe

Navarro, Laetitia M. and Pereira, Henrique M. (2012) *Rewilding Abandoned Landscapes in Europe*, Ecosystems 15: 900-912.

"Conservation is about the past, Rewilding is about the future".
G. Monbiot, 2014.

Laetitia M. Navarro conducted the research following the ideas developed by Henrique M. Pereira, including the CLUE data analysis and the review of biodiversity response to land abandonment and rewilding. LMN wrote the initial draft, which was then edited by HMP.

Resumo:

Durante milénios, a humanidade tem alterado a paisagem, sobretudo através da agricultura. Na Europa, a interacção secular entre o homem e os ecossistemas influenciou fortemente o património cultural. Contudo, por toda a Europa, zonas agrícolas estão a ser abandonadas, particularmente em áreas remotas. A perda destas áreas e a suas consequências para a biodiversidade e serviços de ecossistema tem gerando preocupações, tanto da comunidade científica como do público em geral.

Neste projecto nós pretendemos avaliar em que medida o abandono agrícola pode ser considerado como uma oportunidade para o *rewilding* destes ecossistemas.

Primeiro analisamos as diferentes percepções de agricultura tradicional na Europa e a sua influência nas políticas de gestão do território. Argumentamos que, ao contrário da percepção comum, práticas agrícolas tradicionais não eram ecológicas. Este argumento é ilustrado pelo fato de que a Europa ser actualmente o continente com a menor área de floresta nativa. Além disso, argumentamos que os padrões de vida das populações rurais eram baixos e que os residentes de áreas rurais remotas experimentam um ciclo de declínio, com poucas oportunidades e raramente acesso a amenidades. Sugerimos também que as políticas actuais para manter extensas áreas agrícolas subestimam não só o trabalho humano necessário para sustentar essas mesmas áreas, como as dinâmicas, recentes e futuras, dos factores socioeconómicos responsáveis pelo abandono.

Usando as previsões do modelo CLUE, identificamos as áreas actualmente sob uso agrícola, mas que estão classificadas como naturais em 2030. Estimamos que até 2030, a quantidade de área agrícola que pode ser convertida para área natural pode variar entre 9,9 e 29,7 Mha. Em particular, os *hotspots* de abandono agrícola e *rewilding* foram principalmente identificados em regiões montanhosas e zonas marginais, como os Alpes, os Apeninos, norte de Portugal, ou o noroeste de França e Europa Central.

Estas avaliações confirmam a possibilidade de *rewilding* como uma opção válida

para a gestão do território na Europa, especificamente das áreas que sofreram abandono agrícola. *Rewilding* é uma forma de restauração passiva cujo objectivo final é restaurar ecossistemas auto-sustentáveis. Ao contrário das práticas de conservação centradas na restauração de determinados habitats ou serviços de ecossistema, o *rewilding* foca-se na restauração de processos e na restauração da resiliência do ecossistema.

De seguida, examinamos os potenciais benefícios do *rewilding* para as pessoas e para os ecossistemas. Identificamos mais de uma centena de espécies de mamíferos, aves e artrópodes que podem beneficiar do abandono agrícola e regeneração da floresta. Além disso, argumentamos que, com base nos resultados, pode ser observado um aumento da oferta de alguns serviços de ecossistema, tais como sequestro de carbono e recreação.

Finalmente, discutimos os desafios associados ao *rewilding*, incluindo a necessidade de manter áreas abertas, os riscos de incêndio e os conflitos entre pessoas e vida selvagem. Apesar destes desafios, argumentamos que os decisores políticos devem reconhecer o *rewilding* como uma opção válida para a gestão do território na Europa, particularmente em áreas marginais.

Palavras-chave: Abandono agrícola, alterações do uso do solo, gestão passiva, serviços de ecossistema, *land sharing*, *land sparing*.

Rewilding Abandoned Landscapes in Europe

Abstract: For millennia, mankind has shaped landscapes, particularly through agriculture. In Europe, the age-old interaction between humans and ecosystems strongly influenced the cultural heritage. Yet European farmland is now being abandoned, especially in remote areas. The loss of the traditional agricultural landscapes and its consequences for biodiversity and ecosystem services is generating concerns in both the scientific community and the public. Here we ask to what extent farmland abandonment can be considered as an opportunity for rewilding ecosystems. We analyze the perceptions of traditional agriculture in Europe and their influence in land management policies. We argue that, contrary to the common perception, traditional agriculture practices were not environmentally friendly and that the standards of living of rural populations were low. We suggest that current policies to maintain extensive farming landscapes underestimate the human labor needed to sustain these landscapes and the recent and future dynamics of the socio-economic drivers behind abandonment. We examine the potential benefits for ecosystems and people from rewilding. We identify species that could benefit from land abandonment and forest regeneration and the ecosystem services that could be provided such as carbon sequestration and recreation. Finally, we discuss the challenges associated with rewilding, including the need to maintain open areas, the fire risks, and the conflicts between people and wildlife. Despite these challenges, we argue that rewilding should be recognized by policy-makers as one of the possible land management options in Europe, particularly on marginal areas.

Keywords: farmland abandonment; land-use change; passive management; ecosystem services; land sharing; land sparing.

2.1. Introduction

Deforestation and the loss of natural habitats remain major global concerns. Nonetheless, although scenarios for the next decades project the continuation of these dynamics in tropical ecosystems, the projections made for much of the Northern Hemisphere are quite the opposite (Pereira *et al.* , 2010). In fact, most deforestation in Europe occurred before the industrial revolution (Kaplan *et al.* , 2009), and the amount of forests and scrubland is now increasing following the land abandonment that began in the mid-twentieth century (FAO, 2011), a trend that is expected to continue over the next few decades (van Vuuren *et al.* , 2006).

Natural vegetation recovery is a complex process that occurs during the progressive alleviation of agricultural use (Hobbs & Cramer, 2007; Stoate *et al.* , 2009). This reduction in land-use intensity, including abandonment at the extreme, is, at the local scale, explained by a combination of socio-ecological drivers (MacDonald *et al.* , 2000; Rey Benayas *et al.* , 2007) such as low productivity and aging of the population. These factors interact between them and with the ecological dynamics of succession, creating positive feedback loops, which increase the irreversibility of farmland abandonment in marginal areas, and reduce the effectiveness of subsidies awarded to farmers to halt abandonment (Gellrich *et al.* , 2007; Figueiredo & Pereira, 2011). In Europe, there has been a decline of 17% of the rural population since 1961 (FAOSTAT, 2010). Some parishes of Mediterranean mountain areas have lost more than half of their population in a similar period (Gortázar *et al.* , 2000; Pereira *et al.* , 2005). At the regional scale, the current farmland contraction is best explained by an increase in agricultural productivity and the slowing of population growth in Europe (Keenleyside & Tucker, 2010).

Landowners and managers facing increased agricultural market competition have resorted mostly to one of three active management strategies (Fig. 2.1): intensification, extensification, and afforestation. Intensification is often chosen on the most productive soils and where good conditions exist for mechanization (Pinto-Correia & Mascarenhas, 1999). Extensification consists of obtaining higher productivity by expanding the area of the farm through land consolidation or in developing multiple uses of the land. This has happened in the Montado and Dehesa areas of Portugal and Spain, an agroforestry system that integrates animal production, cork harvesting and cereal cultivation, while hosting high biodiversity and providing recreational and aesthetical benefits (Bugalho *et al.* , 2011). Finally, in some areas with poor

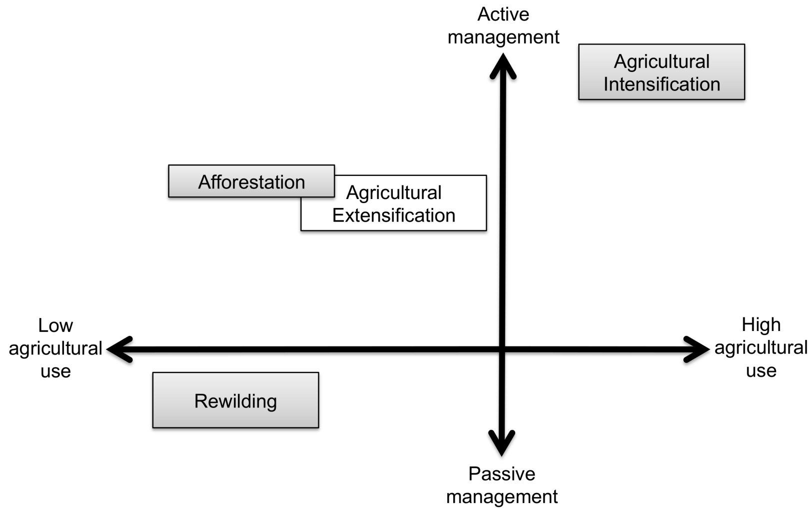

Figure 2.1.: Landscape management strategies plotted against agricultural use intensity and level of management (from active to passive): agricultural intensification, agricultural extensification, afforestation, and rewilding.

farmland soils, the option has been to plant forests, often of fast growing species (Young *et al.* , 2005).

In this article, we discuss a fourth option: rewilding abandoned landscapes, by assisting natural regeneration of forests and other natural habitats through passive management approaches. Rewilding has seldom been considered as a land management policy, as often it faces resistance from both the public (Enserink & Vogel, 2006; Bauer *et al.* , 2009) and the scientific communities (Conti & Fagarazzi, 2005; Moreira & Russo, 2007). Arguments against rewilding include the loss of the traditional agricultural landscape and negative impacts on biodiversity and ecosystem services (for example, Conti & Fagarazzi, 2005). This situation has given rise to a pattern of double standards: developing countries are asked to halt deforestation while some developed countries are actively fighting forest regeneration on their own land (Meijaard & Sheil, 2011).

Here, we critically examine some of the arguments used in support of the maintenance of the traditional landscapes and contrast those arguments with the potential benefits for ecosystem services and biodiversity that could accrue from rewilding. We conclude with an analysis of the main challenges associated with rewilding abandoned landscapes.

2.2. European Landscapes: Examining the Paradigms

The cultural importance of traditional agriculture landscapes has been widely recognized in Europe and the world. As of 2011, 76 of the 936 UNESCO world heritage sites are in the "cultural landscapes" category (http://whc.unesco.org), and 29 of those because of traditional or symbolical agricultural practices. Examples include the "Causses and Cevennes Mediterranean agro-pastoral cultural landscape" in France or the "Mont Perdu" in the Pyrénées. As much as 15 to 25% of the European farmland can be classified as High Nature Value farmland (EEA, 2004). Of the 231 habitat types listed in the European Habitats Directive, 41 are associated with low-intensity agricultural management, including semi-natural grasslands and hay meadows (Halada *et al.* , 2011).

This has lead to a generalized push towards policies embracing the protection of extensive farming systems with the dual-role of protecting biodiversity and ecosystem services. Here we argue that not all socio-ecological aspects of the maintenance of these landscapes have been taken into account because our perceptions of these landscapes have been biased by our own cultural experiences. We question three ideas associated with current policies: (1) the idea that traditional agriculture practices were environmentally friendly; (2) the idea that traditional rural populations lived well; (3) the idea that traditional landscapes can be kept despite the context of recent rural exodus and future socioeconomic trends.

Were traditional agricultural practices environmentally friendly?

In Europe, pre-Neolithic Holocene landscapes can most likely be described as a mosaic of old-growth forest, scrubland, and grasslands, maintained by the grazing of large herbivores and by fire (Svenning, 2002; Vera, 2000, 2009), although the relative amount of open area is debated (for example, Hodder *et al.* , 2009). Later on, and much before the onset of modern agriculture, European inhabitants destroyed most of Europe's forests on usable land. Europe is now the continent with the least original forest cover (Kaplan *et al.* , 2009).

The process of forest clearing might be as old as human's making of tools (Williams, 2000). It started in the Neolithic with the use of fire to open areas for grazing and hunting (see chapter 1). Forest loss was accelerated during Antiquity, when the rise of classical civilizations led to large-scale deforestation (Williams, 2000; Kaplan

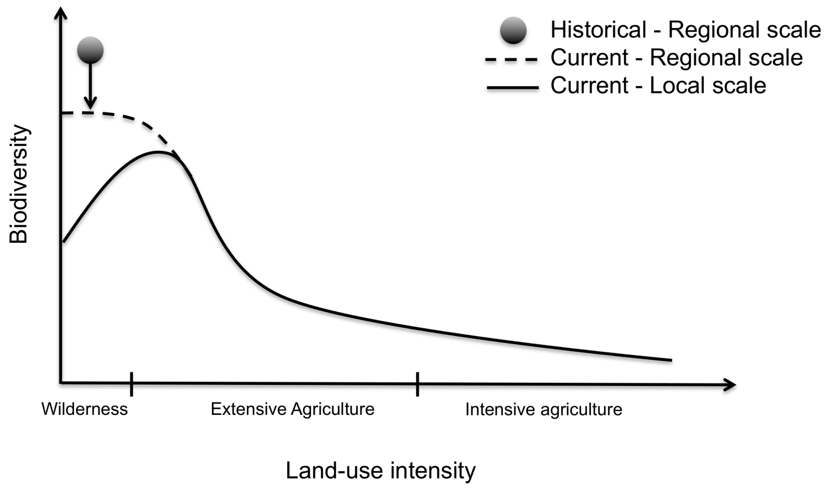

Figure 2.2.: Conceptual representation of the response of current species diversity to land-use intensity at the local and regional scales, and of the hypothetical regional response if Holocene extinctions had not occurred. The response at the local scale is adapted from EEA (2004), whereas the current and historical responses at the regional scale are discussed in the text.

et al. , 2009). After a brief interruption caused by the breakdown of the Roman society, the deforestation trend continued in the Middle Ages (interrupted only by the Black Death), with an estimated loss of 50–70% of the European forest during this period.

Hence humans amplified the disturbance regime of European ecosystems and expanded the open area considerably (Pereira *et al.* , 2012, and see chapter 4), creating and maintaining "traditional" landscapes such as the alpine grasslands (Laiolo *et al.* , 2004), and the agro-silvo-pastoral systems of Mediterranean regions (Blondel 2006). These extensive farming systems have higher species diversity than intensive farming systems (Batáry *et al.* , 2012; Tscharntke *et al.* , 2005), and, at the local scale, often have higher species diversity than non-managed ecosystems and natural forests (Blondel, 2006; Lindborg *et al.* , 2008; Höchtl *et al.* , 2005). Therefore, it has been suggested that biodiversity peaks for low levels of land use associated with these extensive farming systems (Fig. 2.2), following the intermediate disturbance principle (Wilkinson, 1999).

This pattern has been used as an argument to maintain the active management of extensive farmland and halt ecological succession. However at regional scales, this relationship is likely to exhibit a different pattern (Fig. 2.2). The habitat turnover of wild landscapes can be a mosaic of closed forest and open areas, which should accommodate many of the species that can usually be found in extensive farmland habitats. In the early Holocene, the regional diversity of wild landscapes would have been even higher (Fig. 2.2). Several species have now disappeared due to the expansion of human activities, including the auroch (*Bos primeginius*), the Tarpan (*Equus ferus ferus*), or became extinct in most of their former ranges (for example, wisent, *Bison bonasus*).

Deforestation also had important impacts on ecosystem services. In the Mediterranean basin, deforestation is thought to have caused desiccation and soil erosion (McNeely, 1994; Blondel, 2006). In the Middle Ages, timber shortage is likely to have played a role on the impulse to conquer new territories (Farrell *et al.* , 2000). To build naval fleets, countries such as Portugal and Spain had to resort to importing wood from colonies from the sixteenth century on (Devy-Vareta & Alves, 2007). By the end of the nineteenth century, the dimension of the erosion problems in mountain slopes and associated silting in rivers and floods downstream led to large state sponsored afforestation programs in Portugal and Spain.

Did traditional rural populations live well?

For centuries, populations inhabiting marginal agricultural areas organized their lives in a self-sufficient manner (Blondel, 2006). The industrial revolution and the globalization of the food and labor markets brought many of these regions to an economic disadvantage with urban and peri-urban areas: increasing wages associated with economic growth and the low food prices in global markets rendered the low-productivity farmland uncompetitive.

Nowadays, marginal agricultural areas throughout the globe are classified as "poverty traps" where households suffer from scarcity of resources, low return on investment, lack of opportunities, and reduced social services (Conti & Fagarazzi, 2005; Ruben & Pender, 2004). For example, in mountains of Southern Europe, rural populations are constrained by the low productivity of small-scale parcels and the limited opportunities for mechanization and intensification (MacDonald *et al.* , 2000). On average, across European mountain areas, the income per hectare is about 40% lower than in other, non-disadvantaged, areas (809 €/ha vs. 1370 €/ha in European Commission, 2009). The young have limited access to education and employment while the elders experience isolation and difficulties to access services (European

Commission, 2008b). This results in out-migration and aging of the population, leading to an inverted population pyramid. This rural exodus is driven by a "circle of decline" where low population density limits business creation, causing fewer jobs and more out-migrations which, in turn, accentuates the decrease in population density (European Commission, 2008b).

Rural populations still value the quality of their environment and its scenic beauty (Bell *et al.* , 2009; Pereira *et al.* , 2005), but the working conditions in many of these regions have always been difficult. Terraces are some of the most admired cultural landscapes in Mediterranean areas, but locals often use the expression "slavery land" to describe the harshness of the working conditions (Pereira *et al.* , 2005).

Are current efforts to maintain traditional landscapes likely to succeed?

Traditional agricultural practices were characterized by being labor intensive for relatively low agricultural yields (MacDonald *et al.* , 2000; Gellrich *et al.* , 2007). These characteristics played a key role in the demise of many of the traditional practices when labor costs rose due to economic growth, an effect that contributed to and was exacerbated by rural exodus. Large numbers of livestock kept vegetation succession on hold for centuries, but in the past few decades livestock numbers have declined in many of these regions (Cooper *et al.* , 2006). In Europe, the number of livestock (cattle, goats and sheep) declined by 25% between 1990 and 2010 (FAOSTAT, 2010).

Still, recognizing the role of European farmers in maintaining these landscapes (Daugstad *et al.* , 2006), several measures have been implemented to limit farmland depopulation. As part of the European Common Agriculture Policy, Less Favored Areas (LFAs-Regulation 1257/1999) were designated mainly to prevent rural abandonment and maintain cultural landscapes (Dax, 2005; Stoate *et al.* , 2009). LFAs went from representing a third of the European Utilized Agricultural Area (UAA) in 1975 to more than half in 2005 (Dax, 2005; MacDonald *et al.* , 2000). Though the LFA classification often happens to match High Nature Value farming systems and extensive agriculture, it poses no limit to intensification and overgrazing (Dax, 2005).

In the Rural Development Plan for 2007–2013, the payments to farmers in LFAs

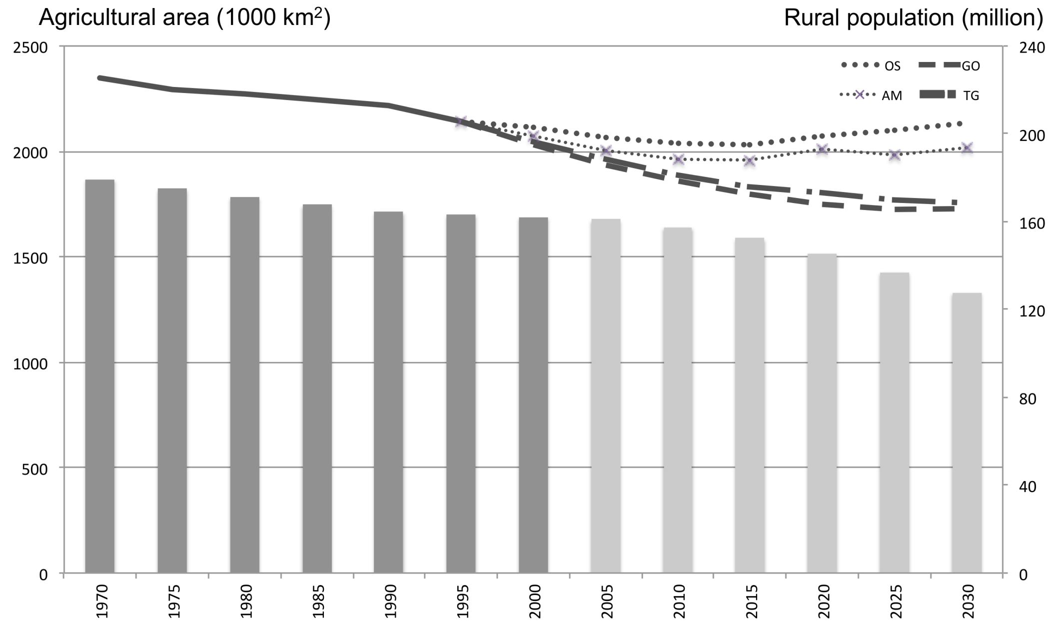

Figure 2.3.: Past and future trends of European agricultural area and rural population. Agricultural area (lines): land-use change predicted in the four scenarios of the Millennium Ecosystem Assessment (van Vuuren *et al.* , 2006). The projections are based on the area of food crops, grass and fodder, and biofuels crops, between 1970 and 2030. OS order from strength, AM adapting mosaic, GO global orchestration, TG techno-garden. Rural population size (bars): historical values (dark gray) and future projections (light gray) (FAOSTAT, 2010; past data for the Baltic countries from http:// www.nationmaster.com).

Region	Variation in the agricultural area	Initial agricultural area (Mha)	Period	Reference
EU15 + Norway and Switzerland	-6% / -10% for cropland -1% / -10% for grassland	142.5 [1]	2000 - 2080	Rounsevell *et al.*, 2006
EU15	+5.5% / -15%	82.5 [1]	2000 - 2030	Eickhout *et al.*, 2007
EU27	-5% / -15%	198	2000 - 2030	Verburg & Overmars, 2009
Europe	-5% / -24%	235	1970 - 2050	MA, 2005
Developed countries[3]	+8% / -20%	183 [2]	2000 - 2050	Balmford *et al.*, 2005

Table 2.1.: Projections of Future Change in the Agricultural Area (Arable Land and Pasture) from Different Studies. (1) Initial agricultural area estimate obtained from FAOSTAT (2010). (2) These values are only for arable land. (3) This study looked at the 23 most important food crops worldwide, corresponding to 44% of the cropland area in developed countries (Balmford *et al.* , 2005).

totaled € 12.6 billion (European Commission, 2011d). Though the sum of these subsidies is substantial at the European scale, at the individual level they might not be enough to maintain young farmers or attract new residents (Cooper *et al.* , 2006), especially in areas where the farm size is small. For example, when considering an average farm size of 23 ha in mountain areas (MacDonald *et al.* , 2000) and an average LFA subsidy of € 100/ha (Dax, 2005), the average payment is of € 2,300 per farm/year. This value can be higher if farmers also adhere to agri-environmental schemes, but overall LFA farmers still have lower incomes (Cooper *et al.* , 2006): the Farm Net Value Added is 13,056 €/ Annual Work Unit in mountain LFAs, 14,174 €/ AWU in other LFAs, and 18,923 €/AWU in non-LFAs (average for the EU25 countries between 2004 and 2005 in European Commission, 2008a).

Hence the decrease in rural populations that started in the 1960s is projected to continue into the next few decades (Fig. 2.3). Future scenarios predict that the contribution of agriculture in regards to GDP and employment in Europe will continue decreasing (Eickhout *et al.* , 2007; Nowicki *et al.* , 2006)and the young generations will keep migrating to the cities, as long as their life quality and income prospects are higher there (European Commission, 2008b; Keenleyside & Tucker, 2010) resulting in the non-replacement of the aging population of European farmers.

Following the decrease in the rural population, agricultural area in Europe is also expected to keep contracting (Fig. 2.3), despite an expected increase in the global demand for agricultural goods, because enough food is obtained either directly by production on competitive land in Europe or elsewhere in the world (Keenleyside & Tucker, 2010). Regionally labeled and organic products could help maintain certain forms of extensive agriculture but this market remains restricted (Strijker, 2005). Projections also take into account an increasing demand in biocrops (Rounsevell *et al.* , 2006; Schröter *et al.* , 2005; Verburg & Overmars, 2009), which can explain a moderate increase in the predicted agricultural area in some scenarios.

The dimension of the agricultural area abandoned or converted into production forest varies widely between scenarios (Tab. 2.1). If we use the intermediate scenarios in Verburg & Overmars (2009), between 10 and 29 million ha of land will be released from agriculture between 2000 and 2030. Areas particularly susceptible to the decline of agropastoral use include semi-natural grasslands and remote or mountainous areas with poor soil quality (Keenleyside & Tucker, 2010; Pointereau *et al.* , 2008; Stoate *et al.* , 2009). Some of these areas are located in Northern Portugal, Northwestern France, the Alps, the Apennines and Central Europe (Fig. 2.4).

2.3. The benefits of rewilding

Defining rewilding

Rewilding is the passive management of ecological succession with the goal of restoring natural ecosystem processes and reducing human control of landscapes (Gillson *et al.* , 2011). Note that although passive management emphasizes no management or low levels of management (for example, Vera, 2009), intervention may be required in the early restoration stages.

In contrast, much of the biodiversity conservation efforts in Europe emphasize active management, by maintaining low-level agricultural practices (Fig. 2.1). Active management also differs in goals, targeting the increase of the abundance of specific taxa or the maintenance of particular habitats, using approaches such as vegetation clearing and construction of artificial habitats, often working against successional processes.

Natural succession on abandoned farmland and pastures often leads to scrubland and sometimes at a later stage, to forest (Conti & Fagarazzi, 2005). Passive forest

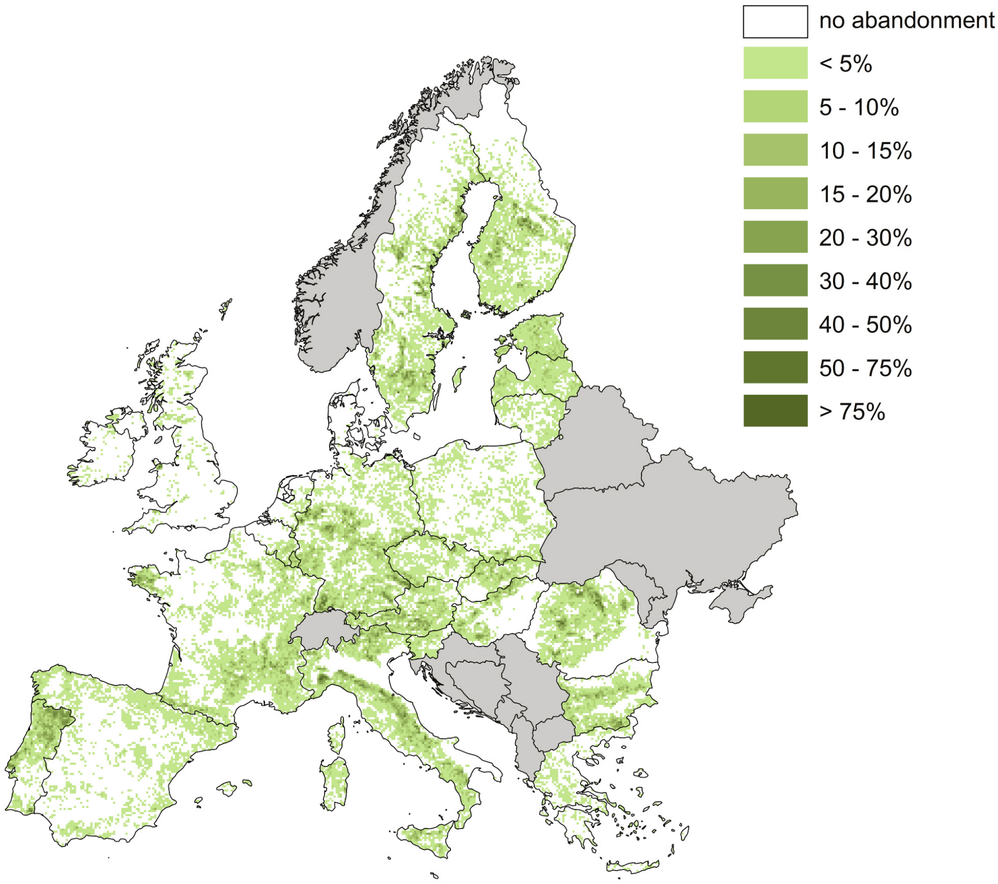

Figure 2.4.: Localization of the hotspots of abandonment and rewilding in Europe. Those hotspots are areas categorized as “agriculture” in 2000 that are projected to become rewilded or afforested in 2030 and that are common to all four scenarios of the CLUE model (Verburg & Overmars, 2009). Hotspots are expressed as a percentage of each 100-km2 grid cell. Agricultural areas correspond to “arable land (non-irrigated)”, “pasture”, “irrigated arable land” and “permanent crops”. Rewilded and afforested areas correspond to “(semi)-natural vegetation”, “forest”, “recently abandoned arable land”, and “recently abandoned pasture land”. Countries in grey have no data.

regeneration restores almost as much forested areas globally as active tree plantation (Rey Benayas & Bullock, 2012). Nonetheless, “wilderness” is not a synonym of “continuous forest” (Sutherland, 2002). The European megafauna played a role in maintaining open landscapes, before being brought to global or local extinction by humans and replaced by domesticated grazers (Bullock, 2009; Vera, 2000; Johnson, 2009; and chapter 4).

This does not mean that rewilding should aim at rebuilding Pleistocene ecosystems, an approach which has been proposed elsewhere (Donlan *et al.* , 2006), but that faces many difficulties (Caro, 2007), including the lack of many of the original keystone species, a different climate, and ecosystems modified locally (for example, changes in soil caused by agriculture) and regionally by humans (for example, the global nitrogen cycle). Instead, the emphasis is on the development of self-sustaining ecosystems, protecting native biodiversity and natural ecological processes and providing a range of ecosystem services (Cramer *et al.* , 2008). These novel ecosystems may be designed to be as similar as possible to some historical baseline in the recent or distant past, but they will often involve the introduction of new biotic elements (Hobbs *et al.* , 2009).

Benefits of rewilding for Biodiversity

Rewilding will cause biodiversity changes with some species declining in abundance, that is, loser species, and other species increasing in abundance, that is, winner species (Russo, 2006; Sirami *et al.* , 2008). We reviewed 23 studies identifying a positive response of species to decreasing human pressure or to restoration of their habitat following land abandonment (see Appendix C)[1]. In total, we identified 60 species of birds, 24 species of mammals, and 26 species of invertebrates that could benefit from farmland abandonment (Table 1 in Appendix C). We also identified 101 species negatively affected by land abandonment (Table 2 in Appendix C), but 13 of those species can be classified as both “winner” and “looser” depending on the study and the region. Much of the agrobiodiversity associated with High Nature Value Farmland will be in the “loosing” category. In contrast, many of the winner species have declined or became functionally extinct in traditional agricultural landscapes, such as large carnivores (Boitani & Linnell, in press). These species will benefit from

[1] see also Merckx (in press); Boitani & Linnell (in press); Cortés-Avizanda *et al.* (in press); Rey Benayas & Bullock (in press) and chapter 4

forest regeneration and the connection of fragmented natural habitats (Keenleyside & Tucker, 2010; Russo, 2006).

Revegetation promotes the increase of the organic matter content and the water holding capacity of soils (Arbelo *et al.* , 2006). This can lead to higher biomasses and densities of earthworms (Russo, 2006) and other invertebrate families (Appendix C, Tab.1.A).

Some forest birds benefit from forest regrowth after farmland abandonment (Pointereau *et al.* , 2008), such as woodpeckers, treecreepers, and tits (Appendix C, Tab.1.B). Some birds of prey have benefited from increases in rodent populations (Pointereau *et al.* , 2008). Perhaps more surprisingly, populations of several bird species of the Eastern European steppe have increased after agricultural activity decline (Hölzel *et al.* , 2002). Some, such as the Little Bustard (*Tetrax tetrax*), have benefited from the tall and dense grassland of the regrown steppes. This contrasts with the concerns that the decrease of open areas in Western Europe is contributing to the decline of steppe species. Therefore the biodiversity consequences of rewilding depend on the geographical context.

Likewise, rural abandonment makes the land suitable for a comeback of large mammals (Appendix C, Tab.1.C). Large grazers are benefiting from the lower hunting pressures that usually accompany abandonment (Breitenmoser, 1998; Gortázar *et al.* , 2000). European carnivore species have been increasing since the 1960s in abundance and distribution, as stable populations of Eastern Europe are naturally recolonizing abandoned landscapes of Scandinavia, the Mediterranean, and the Alps (Enserink & Vogel, 2006; Boitani, 2000; Stoate *et al.* , 2009).

It is also important to consider the trophic interactions between species and the cascading effects driven by rewilding. For example, amphibians and otter (*Lutra lutra*) populations are known to benefit from the restoration of ditches by beavers (*Castor fiber*) in abandoned areas of Eastern Europe (Kull *et al.* , 2004). The presence of lynx in some parts of Switzerland reduced the roe deer and chamois browsing impact by regulating both populations (Breitenmoser, 1998).

Benefits of rewilding for people: ecosystem services

Abandoned farmland is often perceived negatively as it is associated with the perception of unkept land and with the decrease on the economic usability of the land,

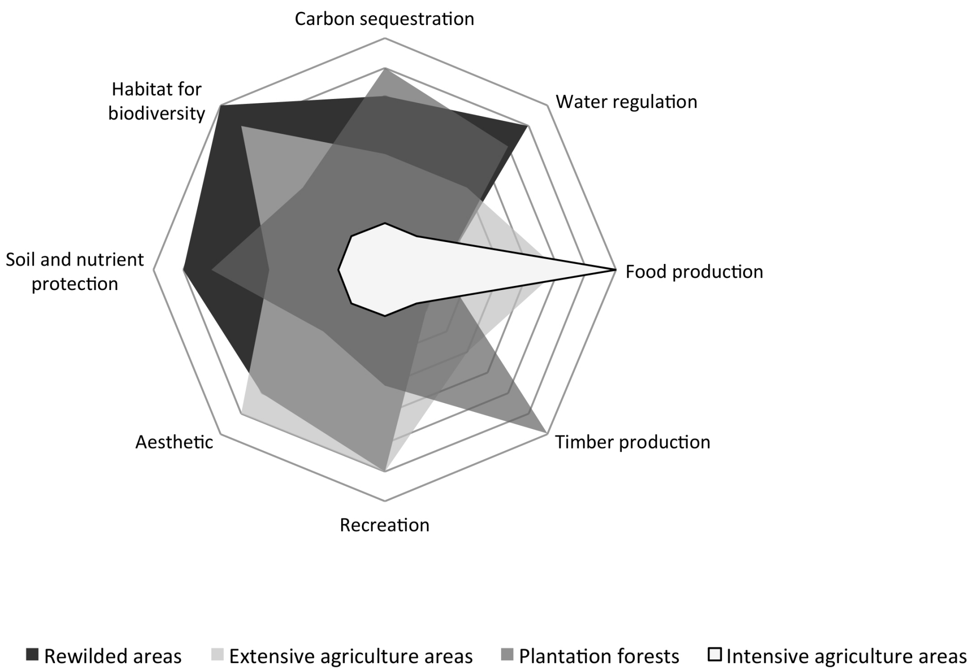

Figure 2.5.: Qualitative assessment of the ecosystem services provided by rewilding, afforestation, extensive agriculture and intensive agriculture in Europe. The relative values given to the provision of each service by the different land management strategies are discussed in the text.

particularly by the rural populations (Bauer *et al.* , 2009; Höchtl *et al.* , 2005). However there are many ecosystems services that are provided by this type of landscapes, particularly indirect and non-use services, which are often disregarded in the process of policy-making (TEEB, 2010).

Rewilded areas can, at the regional scale, provide habitat for biodiversity with conservation results as high or higher than other land management options (Fig. 2.5). This supporting service can lay the foundations for some cultural services (Fig. 2.5), because some of the species benefiting from abandonment are linked with recreation through hunting and tourism (Gortázar *et al.* , 2000; Kaczensky *et al.* , 2004). For instance, in the Abbruze region of Italy, tourism has benefited from the advertisement of the presence of bears and wolves (Enserink & Vogel, 2006). In addition to these direct and indirect use values, the large mammal species brought back by rewilding are amongst the species with highest existence values (Proença *et al.* , 2008).

Forest regrowth promotes carbon sequestration (Kuemmerle *et al.* , 2008). The carbon stock in European forests has grown from 5.3 to 7.7 PgC between 1950 and 1999 (Nabuurs *et al.* , 2003). Nonetheless, active afforestation can potentially yield higher carbon sequestration rates than rewilding by using fast growing species (Fig. 2.5). Natural regeneration allows soil recovery and nutrient availability, though erosion can increase in the first years following abandonment (Pointereau *et al.* , 2008; Rey Benayas *et al.* , 2007). Forests regulate hydrological cycles, particularly in mountain areas (Körner *et al.* , 2005) and water quality is expected to locally improve in abandoned fields (Stoate *et al.* , 2009). Nonetheless, the transition from grassland to forest, a higher water-use system, can reduce the quantity of water (Brauman *et al.* , 2007)). Afforested areas managed for timber provisioning are disturbed both for plantation and management, thus providing qualitatively less water and soil related services than rewilded areas (Fig. 2.5).

Intensive agriculture areas and planted forests are designed to focus on specific provisioning services. Extensive agriculture offers a tradeoff between food provisioning, cultural services, and habitat for biodiversity, whereas rewilding provides a wide range of supporting, regulating and cultural services (Fig. 2.5, and see chapter 5).

The passive management associated with rewilding has much lower maintenance costs than other management options, and therefore significant returns of regulating and cultural services are obtained for limited levels of investment. Still, these services have characteristics of common goods (TEEB, 2010), and therefore are rarely advanta-geous for the individual land-owner. Nonetheless, wilderness is linked to amenity-based growth and attracts urban individuals seeking different environments to both visit and work (Rasker & Hackman, 1996): North American counties favoring wilderness showed faster growth in their employment and income level than counties in which the economy is mainly based on resource extraction.

2.4. The challenges of rewilding

Rewilding as a landscape management option does involve several challenges. Our understanding of those challenges and how they can be overcome depends on the relationship between humans, the landscape and the biodiversity that it sustains.

Conflicts with wildlife

Conflicts occur when wildlife overlaps with human activities such as hunting and farming (Gortázar *et al.* , 2000; Schley & Roper, 2003; Linnell *et al.* , 2000). Those conflicts are age-old in Europe and negative perceptions were transmitted through generations via folklore and tales (Wilson, 2004; Boitani, 2000). Hunting wild species, and particularly carnivores, was socially enforced (Enserink & Vogel, 2006), which led in many cases to their local extinction by the nineteenth century.

Though many European countries have implemented regulations to protect large carnivores, such legislation is not understood and accepted by all (Breitenmoser, 1998). In particular, they accentuate a cleavage in opinions amongst countries and between rural and urban populations (Bauer *et al.* , 2009; Wilson, 2004) the latter being usually more favorable to a wildlife comeback.

The conflicts with carnivores are largely explained by the fact that they prey on domestic animals due to the scarcity of wild prey (Russo, 2006) but also by the loss of traditional livestock-guarding knowledge in several countries (Fourli, 1999; Kaczensky *et al.* , 2004). Nonetheless, the level of depredation of livestock by carnivores is generally low, often less than 10% of their diet (Wilson, 2004). Still, the impact at the level of the livestock owner can be high (Wilson, 2004). To compensate for these impacts, several countries pay for damages caused by wildlife. For bear and wolf damages, an average of € 2 million/year were compensated in Europe between 1992 and 1998 in France, Greece, Italy, Austria, Spain and Portugal (Fourli, 1999) while € 2.15 million were spent in preventive measures.

Large grazers such as deer and wild boars can also cause significant damage to crops, pastures and forest plantations (Goulding & Roper, 2002; Kamler *et al.* , 2010). As for the carnivores, a combination of preventive measures such as electric fencing (Honda *et al.* , 2009) with compensation payments can contribute to decrease the levels of conflict. Fear of attacks on people also play a factor in this conflict, but this often can be improved with better information to the public as there is a correlation between the fear of an animal and a lack of knowledge of its behavior (Decker *et al.* , 2010; Kaczensky *et al.* , 2004).

Limits to ecological resilience

In many regions of Europe, the transition from abandoned to semi-natural land takes less than 15 years, followed by another 15–30 years before reforestation (Cramer *et al.* , 2008; Verburg & Overmars, 2009). Passive regeneration can therefore be a slow process, particularly in a dry environment such as the Mediterranean (Rey Benayas *et al.* , 2008), or when the soils have been modified by past agriculture, that is, the "cultivation legacy" (Cramer *et al.* , 2008), or the "grazing history" (Chauchard *et al.* , 2007). The revegetation also depends on the availability and quality of the native seed bank (Rey Benayas *et al.* , 2008).

If the abandoned land is too degraded assisted regeneration may be needed (Cramer *et al.* , 2008). Active restoration would involve large-scale native trees plantation and tree growth management (Rey Benayas *et al.* , 2008). An intermediate level of intervention involves the creation and management of forest regeneration sources or "woodland islets" (Rey Benayas & Bullock, 2012, in press). Another problem often requiring intervention is the vulnerability of intermediate stages of natural succession to natural perturbations, such as invasive species (Kull *et al.* , 2004; Stoate *et al.* , 2009) and fire (Pausas *et al.* , 2008). Fire is a particularly acute problem as it has impacts not only on biodiversity but also on human health (Proença & Pereira, 2010b). If fire regime is not appropriately managed, frequent fires will favor fire-prone scrubland and halt succession towards forest, in a self-reinforcing feedback loop (Proença & Pereira, 2010a).

One of the strategies to manage fire regimes is to maintain open spaces in the landscape (see chapter 4), minimizing also the impacts of revegetation on species that prefer open areas (Fig. 2.2). This strategy can be implemented by increasing the populations of large herbivores (Hodder & Bullock, 2009; Sutherland, 2002), including reintroduction of extinct species (Svenning, 2002). In the case of species regionally extinct, it is possible to use individuals from other populations. For instance, seven European bison were recently reintroduced in northern Spain, 1,000 years after their extinction (Burton, 2011). A more complex situation occurs with species that are globally extinct, such as wild relatives of some domesticated species. A possible solution is to release into the wild individuals of breeds that are most likely to be successful in replacing the ecological role of their wild ancestors. For instance, Iceland ponies have been released in the former arable fields of the Dutch-Belgian border (Kuiters & Slim, 2003): their grazing favored a dense grass sward and after 27 years open grassland still represented 98% of the area.

Natural colonization of abandoned land by carnivores can also be limited by the availability of prey, as is the case for the Iberian lynx (*Lynx pardinus*) currently negatively affected by the scarcity of rabbits, decimated by diseases (Delibes-Mateos *et al.* , 2008), or as can be expected for some populations of wolves and bears currently preying on livestock (Russo, 2006).

Rewilding may be a future option in areas that are undergoing agricultural development or intensification today. There is currently a debate between land sharing and land sparing approaches to reconcile food production with biodiversity (Phalan *et al.* , 2011). In land sharing, biodiversity conservation and food production goals are met on the same land, with biodiversity friendly agricultural practices and extensive agriculture, whereas in land sparing, land is divided between areas of intensification and of exclusion of agriculture. In practice, it is difficult to determine which is the best option because species respond differently to the alteration of their habitat (Phalan *et al.* , 2011). To maintain future options for rewilding, both land sparing and land sharing are needed. On the one hand, land sharing is essential to limit land degradation and to maintain the appropriate seed bank for future passive revegetation (Rey Benayas & Bullock, in press). On the other hand, land sparing would allow for the conservation of populations of species that are currently in conflict with human activities, making "cohabitation" very difficult.

2.5. Final remarks

Most landscapes are evaluated and protected according to emotional and aesthetic values that societies attribute to them (Gobster *et al.* , 2007; Antrop, 2005) and conservation programs are determined by people's perceptions of what should be preserved (Gillson *et al.* , 2011) and depend on shifting baselines of what nature should be like (Vera, 2009). Thus, the values that Europeans give to farmland and wilderness landscapes are based on tradition and history but also on socio-economic backgrounds (van den Berg & Koole, 2006). Yet, considering that landscapes result from the dy-namic interaction of natural and cultural drivers (Antrop, 2005), they cannot be perceived as anchored in time and we should anticipate occasional changes that will force us to reevaluate their definition.

Rewilding appears to be a viable management option for some of these transitions with important benefits for biodiversity and ecosystem services. At the local scale, some species will decline and other increase, eventually leading to local species diver-sity decreases in some taxa (Fig. 2.2). We lack research studies looking at the regional scale dynamics, but we hypothesize that no significant loss in species diversity is expected as long as mosaics of open spaces and forest are maintained, and that some dimensions of biodiversity may even improve, such as the average size of populations of wild species. At the global scale, many species have already gone extinct and it will be impossible to get them back, but the release into the wild of breeds of some domesticated species may allow recovery of some historical losses (Fig. 2.2). In terms of ecosystem services, rewilding allows for a wide range of regulating and cultural services (Fig. 2.5).

The extent and outcome of rewilding will be heterogeneous across Europe (Fig. 2.4) as different regions will have different departing points of post-farmland abandonment and varying limitations to natural forest regrowth. For example, on some abandoned areas of Southern Europe, the availability of forest tree seed banks can be a limiting factor due to little natural forest left and the frequent fire regime may delay ecological succession. In contrast, the relative scarcity of open areas in much of Northern Europe may render the intensification or reestablishment of natural perturba-tions, such as grazing by large wild herbivores and fire (for example, prescribed burns), priority goals for management. Rewilding can also be considered on available land that does not necessarily result from farmland abandonment, such as national forests previously managed for timber production, decommissioned military areas, salt ponds and other wetlands, thus increasing the level of heterogeneity of European wild landscapes.

From a conservation standpoint, the option between rewilding and active management will depend on the goals and the local context. Active management is likely to be preferred when the goal is to restore specific species or maintain early successional habitats and other habitats associated with human activities. Passive management emphasizes dynamic ecological processes over static patterns of species or habitat occurrence and can be more sustainable in the long term or at large spatial scales.

Despite many benefits, rewilding has been disregarded as a management option until recently. Initiatives such as Rewilding Europe (http:// www.rewildingeurope.com, and see Helmer *et al.* , in press) and the PAN Parks Network (http://www.panparks.org) are now bringing rewilding to the forefront of the discussion of European conservation policies. Rewilding poses many challenges, but those are inherent to the implementation of any restoration plan. In a world wounded by biodiversity loss, farmland abandonment is an opportunity to improve biodiversity in Europe, to study the regeneration of vegetation, and even to test ecological theories (Hobbs & Cramer, 2007). In the end, the question is not whether we prefer a domesticated or a wild European landscape but rather which management options (Fig. 2.1) at each place will be more achievable and sustainable.

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3. A socio-ecological model of sedentarization

Navarro, Laetitia M., Figueiredo Joana, and Pereira, Henrique M. (in prep) *A socio-ecological model of sedentarization: the case of Brandas and Inverneiras in Northern Portugal.*

"Essentially, all models are wrong, but some are useful."

George E. P. Box

Laetitia M. Navarro, Joana Figueiredo and Henrique M. Pereira developed the model. LMN conducted the research and the redaction of the manuscript. All co-authors commented the manuscript.

Resumo:

Apesar de, durante séculos, o sedentarismo ter sido a norma para a maioria da agricultura Europeia, até recentemente algumas comunidades mantiveram comportamentos de migração sazonal. Mais especificamente, na paróquia de Castro Laboreiro, no Parque Nacional da Peneda Gerês (Norte de Portugal), toda a comunidade efectuava migrações entre aldeias de Verão, as *brandas*, e aldeias de Inverno, as *inverneiras*. Como resultado, os residentes utilizavam as áreas circundantes quer das *branda*s quer das *inverneiras* para agricultura. No entanto, desde a década de 1940, a despovoação e a melhoria das condições de vida, especialmente durante o Inverno, tornaram a migração sazonal menos necessária, permanecendo apenas como uma tradição que tem sido progressivamente abandonada.

Nesta secção, utilizamos uma abordagem teórica para investigar a relação entre uma comunidade composta por elementos nómadas e sedentários, e a extensão de área florestada no território.

O modelo socioecológico desenvolvido ilustra a dinâmica interdependente dos sistemas social e ecológico e como estes interagem e levam a mudanças de regime nesta região. A decisão de mudança para um estilo de vida sedentário foi considerada um comportamento colectivo, desencadeado por factores sociais e económicos como a utilidade da terra cultivada, o custo de um estilo de vida nómada e a coesão social dentro da comunidade. Relativamente à parte social do modelo, a terra disponível para cultivo é definida como a terra que não se encontra coberta por floresta. A dinâmica florestal, tanto nas *brandas* como nas *inverneiras*, está ligada ao sistema social através da capacidade de desflorestação dos habitantes.

Este modelo mostra como o conjunto dos sistemas social e ecológico atinge equilíbrios com proporções variáveis de habitantes sedentários, que por sua vez influenciam a quantidade de terra disponível para a regeneração florestal. Podem distinguir-se quatro situações de equilíbrio estável no modelo socioecológico: (1) todos os residentes praticam migração sazonal e usam toda a área quer nas *brandas* quer nas *inverneiras*

para agricultura, pelo que os sistemas não têm qualquer área florestada; (2) todos os residentes praticam migração sazonal, mas deixam alguma área florestada nas *brandas* e/ou nas *inverneiras*; (3) surgem na população residente alguns indivíduos sedentários e áreas variáveis do terreno são usadas para agricultura, deixando algum terreno florestado no sistema; e (4) toda a população cessa a migração sazonal e torna-se sedentária. Neste último caso, o abandono da agricultura nas *inverneiras* causa reflorestação completa, enquanto alguma floresta pode ser mantida nas *brandas*.

Adicionalmente, o modelo socioecológico apresenta algumas mudanças de regime que não seriam observáveis usando apenas o modelo social ou o modelo florestal. Em particular, observamos mudanças de regime entre toda (ou na vasta maioria) a população ser sedentária e toda (ou na vasta maioria) a população ser nómada. Estas alterações no estilo de vida predominante da população também têm repercussões drásticas na cobertura florestal na vizinhança das *brandas* e *inverneiras*.

Transições recentes entre os estilos de vida migrante e sedentário em Castro Laboreiro apoiam este modelo, apesar de uma recente análise detalhada dos padrões de variação no uso dos solos durante os últimos 50 anos mostrar uma dinâmica mais complexa para a cobertura florestal do terreno do que a dinâmica projectada no estudo aqui apresentado.

Apesar disto, a existência de mudanças de regime ilustra a importância de ter em conta as interacções entre as componentes sociais e ecológicas dos ecossistemas. A identificação da existência de pontos de viragem nos ecossistemas é necessária para avaliar a sua resiliência e é particularmente importante quando se considera os serviços disponibilizados pelos ecossistemas, especialmente quando as mudanças de regime resultantes são irreversíveis.

Palavras-chave: migração sazonal, modelo socioecológico, transição crítica, renovação florestal, comportamento colectivo, ponto de viragem.

A socio-ecological model of sedentarization: the case of brandas and inverneiras in Northern Portugal.

Abstract: Though sedentarism has for centuries been the norm for most of the European agriculture, until recently some groups maintained behaviors of seasonal migration. Specifically, in some parishes of Northern Portugal, residents would migrate between summer and winter residences and use both areas for agriculture. The social-ecological model developed here illustrates the linked dynamic of the social and ecological systems and how they interact and lead to regime shifts in this region. The decision to settle was considered as a collective behavior, triggered by social and economic factors while the forest dynamics depend partly on people's ability to use available land. This model shows how the joint social and ecological systems reach equilibria with varying proportions of sedentary residents, which in return influence the amount of land made available for forest regeneration. Such models are important to illustrate regime shifts and to back up numerous empirical observations of linked social-ecological dynamics.

Keywords: seasonal migration, socio-ecological model, critical transition, forest regrowth, collective behavior, tipping point.

3.1. Introduction

The development of agriculture through the domestication of plants and animals some 10 000 years ago marks the transition from hunter-gatherer societies to agricultural societies (Pinhasi *et al.* , 2005) and as such, also triggered the process of sedentarization (Nishida, 2001). Sedentarization has now occurred in most of the world, especially since the advent of land privatization (Nilsson & Fearnside, 2011). Sedentarization has been presented as both a social threshold, with no return to a non-sedentary lifestyle, or as an intermediate stage between periods of higher or lower mobility (for a review see Kelly, 1992). Nowadays, a change in the migration patterns of the remaining nomadic groups, possibly leading to sedentarization, can be triggered by contact with societies that have already settled down (Nilsson & Fearnside, 2011). Though sedentarization and the development of farming might have been considered as a technological improvement in the past (Barker, 2011), a recent study showed that in its early years, agriculture was not more productive than foraging (Bowles, 2011). Moreover, sedentarization involved constraints like parasitism, resource limitation and waste disposal (Nishida, 2001). Religious, social and demographic drivers are likely to play a fundamental role in the development of farming and the underlying process of sedentarization (Barker, 2011; Bowles, 2011). In other words, the decision to settle can be considered as partly triggered by the social environment. As a matter of fact, “Social pressure” has been incorporated in socio-ecological models as major driver of the social dynamic (Satake *et al.* , 2007; Iwasa *et al.* , 2010; Figueiredo & Pereira, 2011).

Historically, in the Mediterranean, when pastoralism was not performed sedentarily, livestock owners would practice either transhumance (i.e. the seasonal displacement of the livestock lead by part of the community) or semi-nomadism (i.e. the seasonal displacement of the entire community between the same areas), the latter involving the movement of entire households (Blondel, 2006). In particular, the inhabitants of some hamlets of Castro Laboreiro in Northern Portugal would practice a seasonal-migration of whole hamlets between the summer residences (Fig. 3.1.A), the *brandas*, and the winter residences (Fig. 3.1.B), the *inverneiras* (Geraldes, 1996; Domingues & Rodrigues, 2008). Since the 1940s, both the rural depopulation and the improvement of living conditions, especially in the winter, made the seasonal migration less of a necessity and more of a tradition, which has been progressively abandoned (Geraldes, 1996; Rodrigues, 2010; van Berkel *et al.* , 2011). This phenomenon was later accentuated by increasing rural outmigration towards cities and emigration to other countries (Daveau, 2003; Graça, 1996; Pereira *et al.* , 2005). Thus, nowadays residents of some communities have settled permanently in the summer villages (*brandas*) while other communities continue to migrate between summer and winter villages (Fig. 3.1.C). The abandonment of traditional agriculture and the associated practices have consequences on the vegetation and the landscape (Cerqueira *et al.* ,

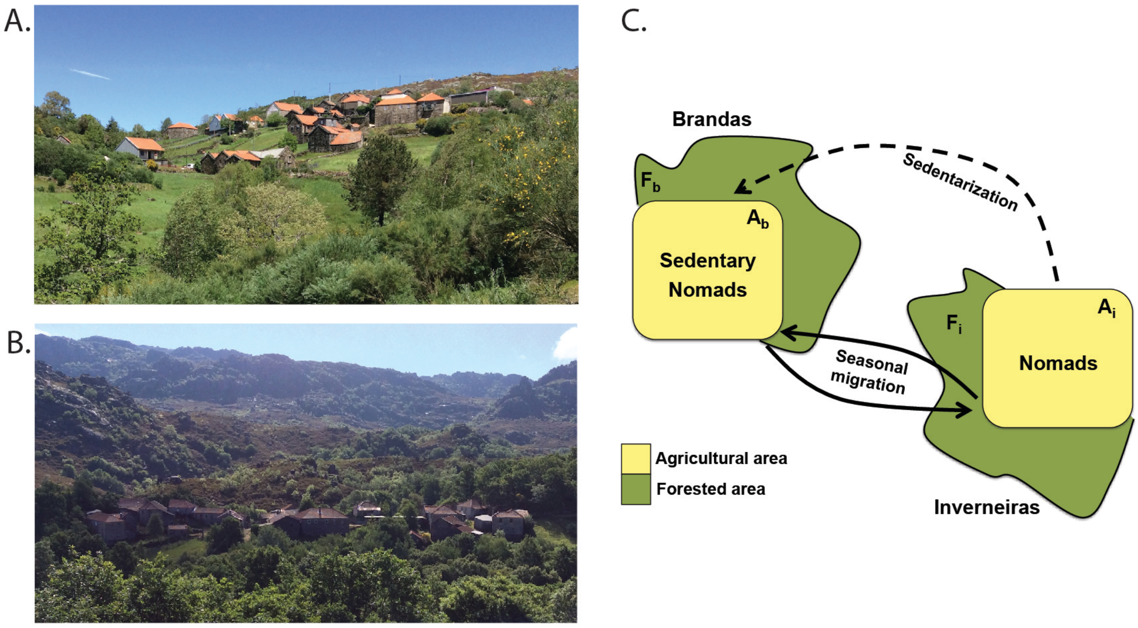

Figure 3.1.: Brandas, Inverneiras, and seasonal migration in Northern Portugal. **A.** Branda, or summer village, of Curral do Gonçalo, in the Castro Laboreiro parish. **B.** Inverneira, or winter village, of Assureira in the Catro Laboreiro Parish. **C.** Schematization of the brandas-inverneiras dynamics. Originally, the entire population migrated seasonally between the summer villages (*brandas*) and the winter villages (*inverneiras*). At one point, some villagers decided to settle in the summer villages. This dichotomy in the social dynamics impacts both the agricultural and forested areas in both "zones". Photos: L.Navarro.

2009) as the abandoned land is prone to natural regeneration (Cramer *et al.* , 2008; Verburg & Overmars, 2009; Keenleyside & Tucker, 2010).

In order to effectively study the reciprocal influence of ecological and social dynamics and to identify feedbacks and thresholds, several authors emphasized the need to couple the natural and human systems into socio-ecological models (Peterson, 2000; Liu *et al.* , 2007; Carpenter *et al.* , 2009; Milne *et al.* , 2009). Here, we developed a model describing the dynamics of the ecological systems (i.e. forests/farmlands ecosystems) linked with a social system (i.e. sedentarization against nomadism) and their integration in a socio-ecological model. These models are based on the farmland-forest socio-ecological model developed by Figueiredo & Pereira (2011). Specifically, we use this model with the case of seasonal migrations (i.e. number of individuals with a migratory and a sedentary lifestyle and the transitions between both) and apply it to the case of migrations between brandas and inverneiras observed in some locations of Northern Portugal. The aim of this model is firstly to identify the social thresholds driving either the abandonment or the resumption of seasonal migrations, and secondly to understand the relationship between those social patterns and the forest dynamics.

3.2. Ecological model

Farmland-forest dynamics depend on both forest growth and deforestation rate (Figueiredo & Pereira, 2011). Forest areas grow logistically with ε being the forest rate of increase and T the carrying capacity of the system (corresponding to the maximum forest area). Deforestation (or conversely, the rate of increase of the farmland area), will depend on the number of residents, R, and their ability to conquer land from the forest through logging, λ (the deforestation ability per resident). We assume that the total deforestation rate per resident is proportional to the amount of forest left: as forest declines, the remnant patches will be more difficult to access and harvest. The state variables of the forest-farmland system are the forest F, and farmland areas, A, with $A = T - F$. Hence, the forest area dynamics are given by:

$$\frac{dF}{dt} = \varepsilon F \left(1 - \frac{F}{T}\right) - \lambda R F \tag{3.1}$$

This forest-farmland system has two equilibria:

$\hat{F} = 0$ and $\hat{F} = \frac{T}{\varepsilon}(\varepsilon - R\lambda)$.

Based on the stability analysis (see Figueiredo & Pereira, 2011), if $R\lambda > \varepsilon$, the equilibrium $\hat{F} = 0$ is stable, that is, when the deforestation rate exceeds the forest rate of increase the forest tends to disappear. If $\varepsilon > R\lambda$, $\hat{F} = \frac{T}{\varepsilon}(\varepsilon - R\lambda)$ is the stable equilibrium, that is, when the forest rate of increase exceeds the deforestation rate, the forest area will (i) increase with the forest rate of increase, ε, (ii) decrease linearly with the residents' deforestation ability, λ, (iii) increase linearly with the decrease of number of residents until dominating the landscape when humans completely abandon the area (when R = 0, $\hat{F} = T$).

Forest dynamic applied to the sedentarization context

The population using the *inverneiras* corresponds to the fraction of the population performing seasonal migration, the "seasonal nomad" individuals (hereafter nomads), N, while the population in the *brandas* is composed of both nomads and sedentary residents, S. Hence, the dynamics of the forest-farmland systems, in the *brandas* and in the *inverneiras*, are given by:

$$\frac{dF_i}{dt} = \varepsilon_i F_i \left(1 - \frac{F_i}{T_i}\right) - \lambda_i N F_i \tag{3.2}$$

$$\frac{dF_b}{dt} = \varepsilon_b F_b \left(1 - \frac{F_b}{T_b}\right) - \left(\lambda_{bS} S + \lambda_{bN} N\right) F_b \tag{3.3}$$

The subscripts b and i stand for *brandas* and *inverneiras*, respectively (for a descriptive list of the parameters, see Appendix D). The equilibrium and stability analysis of the forest model applied to the sedentarization context is the same as in the general framework presented in the previous section. In particular, the dynamic for the forest in the *inverneiras* is at equilibrium when $F_i = 0$ or $F_i = \frac{T_i\varepsilon_i - RT_i\lambda_i + ST_i\lambda_i}{\varepsilon_i}$. In the *brandas*, the equilibrium of the forest dynamic is reached when $F_b = 0$ or $F_i = \frac{T_b(\varepsilon_b - R\lambda_{bN} + S\lambda_{bN} - S\lambda_{bS})}{\varepsilon_b}$.

3.3. Social model

The social model describes the dynamics of the sedentary individuals, S and the number of nomad individuals, N, within a resident population of constant size, R. Here, we only describe the dynamics of the number of sedentary individuals, S, since $S + N = R$.

The decision to settle can be seen as a collective behavior, which can be described with threshold models (Granovetter, 1978): individuals are offered two alternatives, the threshold for decision depends on the number of individuals that already chose a given alternative, and corresponds to a point where the net benefits of that decision exceed the costs for that individual. Collective behaviors are commonly used to describe and model human actions such as participation to riots, to strikes and to political revolutions (e.g. Kuran, 1989); voting patterns, the diffusion of rumors, of innovations and of new technologies (e.g. Toole *et al.* , 2012). Moreover, social experiments can demonstrate the tendency of humans to normalize their behavior within their group (e.g. Goldstein *et al.* , 2008; Schultz *et al.* , 2007).

As a result, we can expect that at any time step, the probability to become sedentary will depend on the proportion of sedentary residents in the population. Note that the feeling of being engaged with the rest of the community is more likely to be dependent on the relative proportion of neighbors with a similar behavior than on the absolute number of neighbors (e.g. ten individuals becoming sedentary in a very small community have a much greater impact than in a very large community). Thus, individuals settle at a rate given by:

$$\frac{dS}{dt} = f\left(\frac{S}{R}\right) \tag{3.4}$$

where f is a function describing the rate of sedentarization based on the proportion of the population that already decided to stop seasonal migration. This social process follows the same logic as the process of migration between rural and urban systems described by Figueiredo & Pereira (2011). Thus, each individual from the population chooses between a nomadic or a sedentary lifestyle and the decision is based on a personal threshold: if the proportion of neighbors that have already become sedentary is higher than that threshold, the individual also becomes sedentary. The model assumes that the distribution of personal thresholds has a logistic distribution. Therefore the cumulative distribution function of the logistic distribution, $CDF\left(\frac{S}{R}\right)$, gives the cumulative proportion of individuals of the population that have a threshold less than or equal to $\frac{S}{R}$,

$$CDF\left(\frac{S}{R}\right) = \frac{1}{1 + \exp\left(\frac{\mu - \frac{S}{R}}{\delta}\right)} \tag{3.5}$$

where μ (location parameter) is the mean threshold in the population and δ is a scale parameter ($\delta > 0$) which represents the strength of personal connections (social bonding) in the population. A smaller δ represents a population where individuals have similar thresholds (i.e. with a small variance in μ), and therefore, a greater synchronism of settlement (Granovetter, 1978). In this case the value of the cumulative distribution function rapidly reaches a maximum (Fig. 3.2.A). This means that with high social bonding (low δ), if the number of sedentary individuals increases, the probability that individuals reach their threshold increases as well, and rapidly. In the case of a population with low social bonding, this phenomena is much more gradual and in some cases ($\mu > 0.5$) the probability that individuals reach their threshold is never maximal, even if all neighbors become sedentary (Fig. 3.2.B).

The change in the number of sedentary individuals per unit time is given by the probability to take a decision to settle per unit time (ω) multiplied by the cumulative proportion of individuals from the population that have a threshold less than or equal to $\frac{S}{R}$, $CDF\left(\frac{S}{R}\right)$, minus the proportion of individuals from the population that have already settled, $\frac{S}{R}$. This value must be multiplied by the total size of the resident population. Therefore,

$$\frac{dS}{dt} = \omega \times \left[CDF\left(\frac{S}{R}\right) - \frac{S}{R}\right] \times R = \omega \left(\frac{R}{1 + \exp\frac{\left(\mu - \frac{S}{R}\right)}{\delta}} - S\right) \tag{3.6}$$

The equilibrium of this equation cannot be theoretically determined, and has to be estimated numerically (see Figueiredo & Pereira, 2011 for details of the equilibrium analysis). When members of the resident population have dissimilar thresholds (large values of δ), the number of sedentary individuals in equilibrium, $\hat{S}$, decreases with increasing individual threshold μ. When individuals have similar thresholds (small values of δ), the social system has several equilibria. When μ is small, there is one stable equilibrium, $\hat{S} = R$, meaning that given a low threshold in the population, individuals will settle. When μ is high, there is one stable equilibrium, $\hat{S} = 0$, meaning that the entire population practices seasonal migration. When μ has an intermediate value, the number of sedentary individuals in equilibrium will depend on the initial number of sedentary individuals: if $S < \frac{R}{2}$, the number of sedentary individuals at equilibrium will be close to $\hat{S} = 0$; if $S > \frac{R}{2}$, the number of sedentary

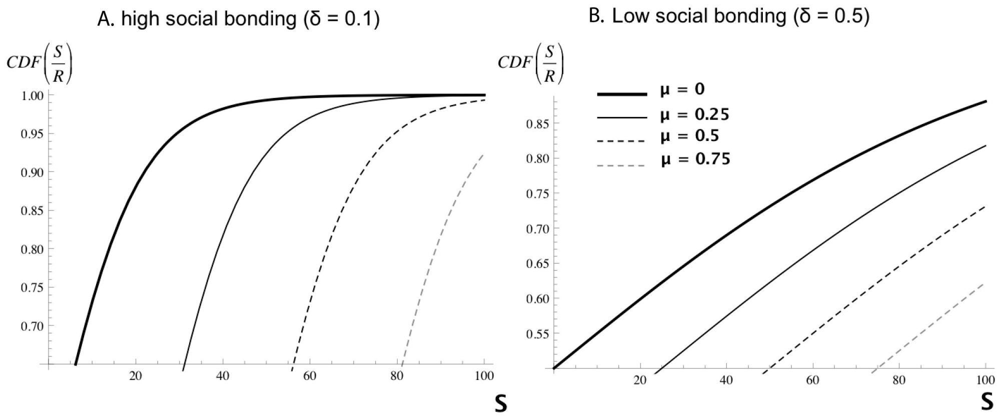

Figure 3.2.: Cumulative distribution function of the logistic distribution applied to the social dynamic of the model. The cumulative distribution function is calculated as a function of μ, the individual threshold, S the number of sedentary individuals in the population, and δ, representing the social bonding. It represents the proportion of the population R that has a threshold inferior or equal to the population that already decided to sedentarize, $\frac{S}{R}$.

individuals at equilibrium will be close to $\hat{S} = R$. In the case of a population with a high social bonding (i.e. a small δ), the dynamics shows a case of hysteresis, where a return to the initial conditions cannot simply be achieved by restoring the values of the parameters to their original state (Figueiredo & Pereira, 2011).

Farmland utilities and the process of sedentarization

Since μ represents the threshold triggering the decision to change one's lifestyle in the population, it will depend on the respective utilities of both the sedentary and nomadic ways of life. The model considers that people are more likely to stop seasonal migration once the utility of the "migrant lifestyle" becomes lower than the utility of the "sedentary lifestyle". We consider that the farmland utility is determined by the agricultural activity, $A_x h_x$, where A_x represents the farmland area of type x and h_x the utility obtained per unit of agricultural area of type x. The utility of the "nomadic" life style, U_N, is given by the utility of the winter farms, plus the utility of the summer farms, to which we must add utilities from other sources, σ. There is also a "cost of seasonal migration", c, associated to the movement of animals and people from the summer to the winter villages, and which decreases the utility derived from the summer farms. Hence,

$$U_N = \frac{A_i h_i}{R - S} + (1 - c)\,\frac{A_b h_b}{R} + \sigma \tag{3.7}$$

The utility of a sedentary lifestyle is given by the utility of using the summer residences, the brandas, year-round, plus the utility derived from other sources.

$$U_S = \frac{A_b h_b}{R} + \sigma \tag{3.8}$$

Each individual will decide to settle if the proportion of sedentary individuals equals or exceeds its personal threshold. Since μ represents the proportion of sedentary residents, it can only take values between 0 and 1. We assume that the threshold equals half of the population having already settled when the utilities of both lifestyles are equal, that is $\mu = 0.5$ when $U_N = U_S$ and

$$\mu = \frac{\frac{A_i h_i}{R-S} + \frac{(1-c)A_b h_b}{R} + \sigma}{2\left(\frac{A_b h_b}{R} + \sigma\right)} \tag{3.9}$$

Thus, the social dynamic is given by:

$$\frac{dS}{dt} = \omega \left[\frac{R}{1 + \exp \frac{\left[\frac{\frac{A_i h_i}{R-S} + (1-c)\frac{A_b h_b}{R} + \sigma}{2\left(\frac{A_b h_b}{R} + \sigma\right)} - \frac{S}{R}\right]}{\delta}} - S \right] \tag{3.10}$$

The dynamic of sedentarization is then determined by the interaction between the social bonding and the balance of the utilities of the nomadic and sedentary lifestyles. Decreasing the utility of the sedentary lifestyle in a population with high social bonding causes a sharp decline in the sedentary population (Fig. 3.3.A). The sedentary population will hardly recover, even if the utility of the sedentary lifestyle is increased to a level equivalent to the utility of the nomadic lifestyle. The same events in a population with low social bonding are less pronounced. This difference between low and high social bonding is also observed when looking into the response of the sedentary population to a variation in the cost of the seasonal migration (Fig. 3.3.B). Even with no cost for nomadism, some sedentary individuals will appear in a population with low social bonding whereas, with a tight social frame, the cost of nomadism must be close to maximum for sedentary individuals to appear in a population.

3.4. Socio-ecological model

The equations of the socio-ecological (SES) model for the farmland-forest ecosystem result from the combination of the equations of the ecological and social models (Eq.3.2, 3.3, 3.10). In the SES model the number of nomad individuals, N, and

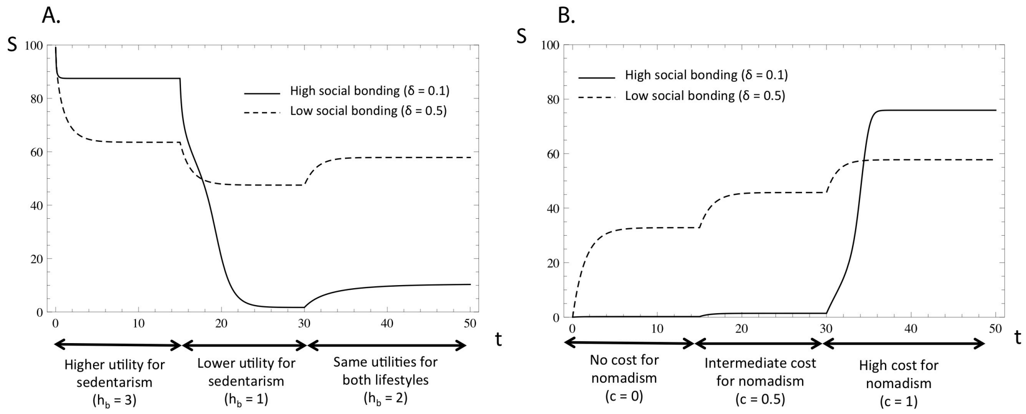

Figure 3.3.: Dynamic of the sedentary population. **A.** Sedentary population with varying values of the utility of the farmland area in the brandas over time (h_b = 3, 1 and 2 ha^{-1} respectively for t < 15 years, 15 < t < 30 years and t > 30 years) in populations with both a high and a low social bonding ($\delta = 0.1$ or $\delta = 0.5$). **B.** Sedentary population with varying values of the cost of seasonal migration over time (c = 0, 0.5 and 1 respectively for t < 15 years, 15 < t < 30 years and t > 30 years) in populations with both a high and a low social bonding ($\delta = 0.1$ or $\delta = 0.5$). For both a) and b) A_i= 20ha, A_b = 75 ha, R = 100 individuals, $S(0)$ = 99 individuals, h_i = 2 ha^{-1}, $\sigma = 1$, $c = 0.9$, $\omega = 1$ ind^{-1}years^{-1}.

the farmland areas, A_i and A_b, are no longer parameters, but state variables. The number of nomad individuals N using the inverneiras is expressed as a function of the number of sedentary individuals in the population, $R - S$. The farmland areas, A_i and A_b, are represented by the portion of the total area not occupied by forest in each location and are given by $(T_i - F_i)$ and $(T_b - F_b)$, respectively. Therefore, the equations of the socio-ecological model (SES model) are:

$$\frac{dF_i}{dt} = \varepsilon_i F_i \left(1 - \frac{F_i}{T_i}\right) - \lambda_i \left(R - S\right) F_i \tag{3.11}$$

$$\frac{dF_b}{dt} = \varepsilon_b F_b \left(1 - \frac{F_b}{T_b}\right) - \left(\lambda_{bS} S + \lambda_{bN} \left(R - S\right)\right) F_b \tag{3.12}$$

$$\frac{dS}{dt} = \omega \left[\frac{R}{1 + \exp \frac{\left[\frac{\frac{(T_i - F_i)h_i}{R-S} + (1-c)\frac{(T_b - F_b)h_b}{R} + \sigma}{2\left(\frac{(T_b-F_b)h_b}{R} + \sigma\right)} - \frac{S}{R} \right]}{\delta}} - S \right] \tag{3.13}$$

The forested area in the *brandas*, F_b, and in the *inverneiras*, F_i, along with the number of sedentary individuals, S, may now vary over time, affecting the dynamic of the social and the ecological systems by reciprocally influencing each other.

Equilibrium and stability analysis

As the equilibria for these equations cannot be calculated analytically, numerical examples were used to characterize the effect of the model's parameters. We plotted the zero isoclines of the socio-ecological model (see Fig. 3.4 for an example). The points of intersection of the isoclines of the three models are the equilibria of socio-ecological model (Tab. 3.1 and Tab. 3.2). The sum of the vectors of change of the models indicates whether the equilibrium is stable or unstable: if the vectors point to the intersection, the equilibrium will be stable; if they point in opposite directions, the equilibrium will be unstable. The stability of the equilibria was also confirmed by looking at the eigenvalues of the Jacobian matrix of the system at equilibria.

We have numerically examined two values of social bonding, high ($\delta = 0.1$ in Tab. 3.1) and low ($\delta = 0.5$ in Tab. 3.2), two values of forest growth rate, low ($\varepsilon = 0.1$) and high ($\varepsilon = 4$), and two values of the utility of the farmland in *brandas*, low ($h_b = 1$) and high ($h_b = 3$). In the case of a population with high social bonding, low forest growth and low utility in the *brandas* (Tab. 3.1), one stable equilibrium occurs when all the variables have values close to 0 ($\hat{F}_i \approx 0$, $\hat{F}_b \approx 0$, and $S \approx 0$), i.e. when all are nomads and use all the area in both *brandas* and *inverneiras* for agriculture. If forest growth is high (Tab. 3.1) then some forest persists in the *brandas*, $\hat{F}_b = \frac{T_b(\varepsilon_b - R\lambda_{bN} + S\lambda_{bN} - S\lambda_{bS})}{\varepsilon_b}$, and in the *inverneiras*, $\hat{F}_i = \frac{T_i\varepsilon_i - RT_i\lambda_i + ST_i\lambda_i}{\varepsilon_i}$.

If the utility in the *brandas* is high, a second stable equilibrium occurs for low forest growth, with all the population becoming sedentary ($\hat{S} \approx 100$), and all the forest

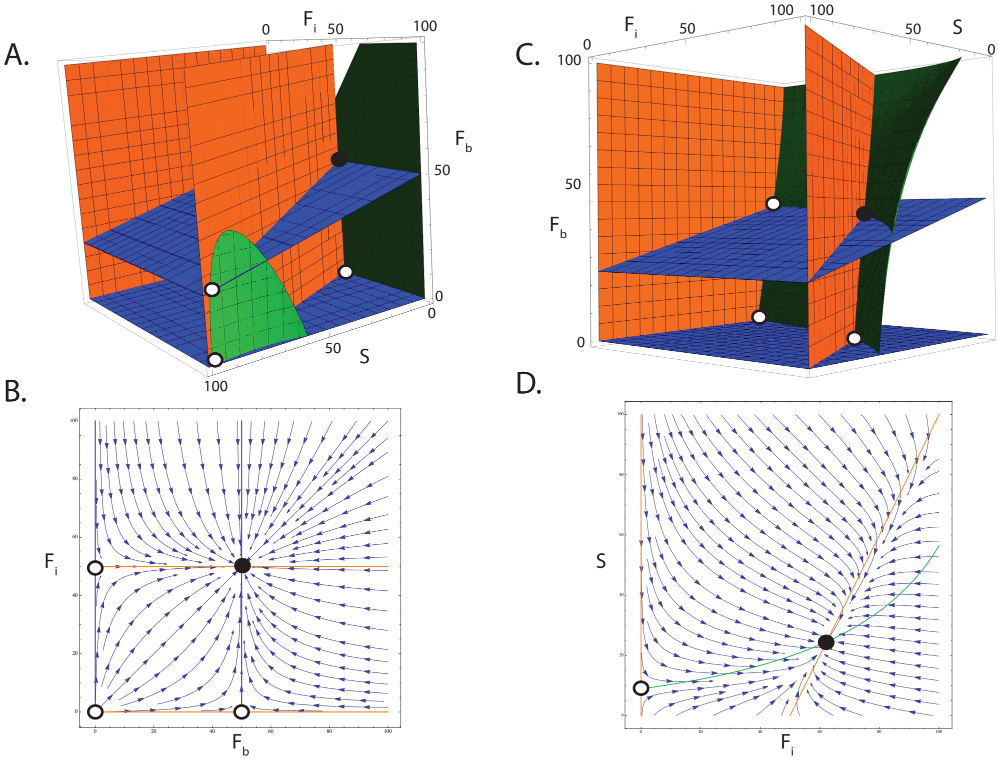

Figure 3.4.: Examples of 3D and 2D plots of the phase portraits of the socio-ecological dynamic.
A. and **B.** Socio-ecological system in the case of high social bonding ($\delta = 0.1$), high forest growth ($\varepsilon_i = \varepsilon_b = 4$) and lower utility in the brandas ($h_b = 1$ ha^{-1}).
B. is the phase portrait of the model for $\hat{S} \approx 0$.
C. and **D.** Socio-ecological system in the case of high low social bonding ($\delta = 0.5$), high forest growth ($\varepsilon_i = \varepsilon_b = 4$) and higher utility in the brandas ($h_b = 3$ ha^{-1}).
D. is the phase portrait of the model for $\hat{F_b} \approx 44$.
For all plots: $\omega = 1$ ind^{-1} years^{-1}, $T_i = 100$ ha, $T_b = 100$ ha, $R = 100$ ind., $h_i = 2$ ha^{-1}, $\sigma = 1$, $c = 0.9$, $\lambda_i = \lambda_{bN} = 0.02$ and $\lambda_{bS} = 0.03$. A black dot represents a stable equilibria, a white dot an unstable one.

High Social Bonding (δ =0.1)

	Low utility in ***brandas*** (h_b = 1)	High utility in ***brandas*** (h_b = 3)
Low forest growth ε = 0.1	$\hat{F}_b \approx 0$ and $\hat{S} \approx 0$ and $\hat{F}_i \approx 0$	$\hat{F}_b \approx 0$ and $\hat{S} \approx 0$ and $\hat{F}_i \approx 0$ $\hat{F}_b \approx 0$ and $\hat{S} \approx 100$ and $\hat{F}_i \approx 100$
High forest growth ε = 4	$\hat{F}_b \approx 50$ and $\hat{S} \approx 0$ and $\hat{F}_i \approx 50$	$\hat{F}_b \approx 50$ and $\hat{S} \approx 0$ and $\hat{F}_i \approx 50$

Table 3.1.: Stable equilibria with high social bonding ($\delta = 0.1$), varying values of the forest growth in both *brandas* and *inverneiras* (low $\varepsilon_i = \varepsilon_b = 0.1$, and high $\varepsilon_i = \varepsilon_b = 4$) and of the utility of the land in the *branda*s, (lower utility $h_b = 1$ ha^{-1}and higher utility $h_b = 3$ ha^{-1}) with $\omega = 1$ ind^{-1} years^{-1}, $T_i = 100$ ha, $T_b = 100$ ha, $R = 100$ ind., $h_i = 2$ ha^{-1}, $\sigma = 1$, $c = 0.9$, $\lambda_i = \lambda_{bN} = 0.02$ and $\lambda_{bS} = 0.03$.

Low Social Bonding (δ =0.5)

	Low utility in ***brandas*** (h_b = 1)	High utility in ***brandas*** (h_b = 3)
Low forest growth ε = 0.1	$\hat{F}_b \approx 0$ and $\hat{S} \approx 4$ and $\hat{F}_i \approx 0$	$\hat{F}_b \approx 0$ and $\hat{S} \approx 18$ and $\hat{F}_i \approx 0$
High forest growth ε = 4	$\hat{F}_b \approx 48$ and $\hat{S} \approx 7$ and $\hat{F}_i \approx 54$	$\hat{F}_b \approx 44$ and $\hat{S} \approx 24$ and $\hat{F}_i \approx 62$

Table 3.2.: Stable equilibria with low bonding ($\delta = 0.5$), varying values of the forest growth in both *brandas* and *inverneiras* (low $\varepsilon_i = \varepsilon_b = 0.1$, and high $\varepsilon_i = \varepsilon_b = 4$) and of the utility of the land in the *branda*s, (lower utility $h_b = 1$ ha^{-1} and higher utility $h_b = 3$ ha^{-1}) with $\omega = 1$ ind^{-1} years^{-1}, $T_i = 100$ ha, $T_b = 100$ ha, $R = 100$ ind., $h_i = 2$ ha^{-1}, $\sigma = 1$, $c = 0.9$, $\lambda_i = \lambda_{bN} = 0.02$ and $\lambda_{bS} = 0.03$.

remaining in the *inverneiras* ($\hat{F}_i \approx 100$). In the case of a population with low social bonding (Tab. 3.2), a population of sedentary inviduals is present for each stable equilibrium, though in low numbers ($\hat{S} < 25$). If the forest growth is low, there is no forest left in the system at equlibrium, whereas with a high forest growth, forested area remains in both *brandas* and *inverneiras* at equilibrium.

Examining the behavior of the socio-ecological models for a range of parameters we find that lowering the social bonding strongly decreases the likelihood of having a population composed only of sedentary individuals and therefore a complete reforestation in the *inverneiras* is not likely to occur.

3.5. Regime shifts

The interactions between the different elements of socio-ecological models are not linear and are governed by the system's thresholds (Liu *et al.* , 2007). The states of the ecological and social regimes will shift, either suddenly or gradually, and influence each other once those thresholds are crossed (Walker & Meyers, 2004; Kinzig *et al.* , 2006; Carpenter *et al.* , 2009).

When looking into the dynamic of the social system only, with a static forested area, a sedentary population is established when the utility in the *brandas*, h_b is higher than the utility in the *inverneiras*, h_i, at least when the social bonding is strong. In a scenario where the utility in the *brandas* then decreases gradually through time, this sedentary population will drop completely and all the population is assumed to resume a behavior of seasonal migration (Fig. 3.5.A).

The same scenario ran with the socio-ecological model shows a different response of the sedentary population. Specifically, once it has reached a maximum, whether the utility in the *brandas* decreases or not, all the residents remain sedentary (Fig. 3.5.B). As a result, the forest area also increases to a maximum in the *inverneiras*, which are left to forest regeneration after the abandonment of seasonal migration. Simultaneously, all the area in the *brandas* is devoted to agriculture, leaving no room to the forest. In this particular case, the socio-ecological model shows a case of irreversibility of the system that is not observed when only the social dynamics are examined.

3.6. Discussion

This socio-ecological model of sedentarization illustrates the possible transition from a population performing seasonal migration to a sedentary rural population. We applied it to the case of the *brandas/inverneiras* system found in Castro Laboreiro, Northern Portugal. In particular, the SES model shows how the forest and human dynamics interact and generate thresholds and regime shifts that would not be observed with a forest or social model only.

The forest dynamics are influenced by the farming activities of the human population that cause deforestation. Reciprocally, the ability of forest to re-conquer abandoned farmland can influence the likelihood of residents to farm a given area.

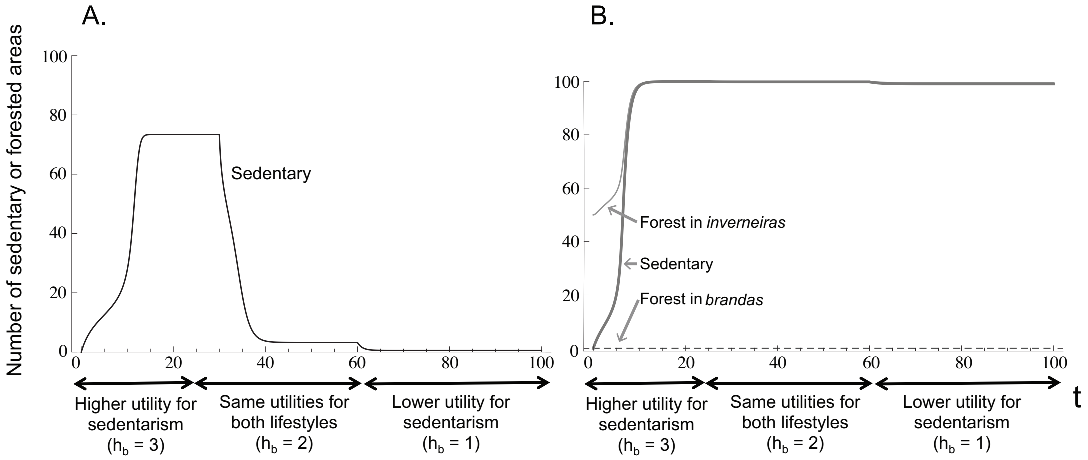

Figure 3.5.: Comparison of the social only and socio-ecological dynamics with high social bonding and varying utility of the sedentary lifestyle.
A. The forests dynamics are static and the number of sedentary individuals (S) depends only on the social dynamic with h_b = 3, 2 and 1 ha^{-1}respectively for t < 25 years, 25 < t < 60 years and t > 60 years; S(0) = 0 ind., A_i = 50 ha, A_b = 100 ha, R = 100 ind., h_i = 2 ha^{-1}, $\sigma = 1$, $c = 0.9$, $\omega = 1$ ind^{-1} years^{-1} and $\delta = 0.1$.
B. The forests and sedentary population dynamics are linked in the socio-ecological model with h_b = 3, 2 and 1 respectively for t < 25 years, 25 < t < 60 years and t > 60 years; S(0) = 0 ind., $F_i(0)$ = 50 ha and $F_b(0)$ = 0 ha; $\omega = 1$ ind^{-1} years^{-1}, T_i = 100 ha, T_b = 100 ha, R = 100 ind., h_i = 2 ha^{-1}, $\sigma = 1$, $c = 0.9$, $\lambda_i = \lambda_{bN} = 0.02$ and $\lambda_{bS} = 0.03$, $\varepsilon_i = \varepsilon_b = 4$ and $\delta = 0.1$.

This is consistent with empirical studies that observe more regeneration of vegetation clearings when human populations practice migration (nomadism) than when they are sedentary (e.g. Nilsson & Fearnside, 2011). Moreover, historically, sedentarization of the early European societies has been linked with extensive deforestation (Williams, 2000). Still, the case of migration presented in this model, where the residents would move entire households and village facilities from one site to another is quite unique. A more general approach to the modeling of sedentarization would involve the migration between sites that are different each year.

Four situations can be described when looking into the stable equilibria of the socio-ecological model: (1) all the residents practice seasonal migration and use all the area in both *brandas* and *inverneiras* for agriculture, hence the systems have no forested area at all; (2) all the residents practice seasonal migration but they leave some forested area in the *brandas* and/or in the *inverneiras*; (3) some sedentary individuals appear in the resident population and various amount of area in the *brandas* and the *inverneiras* are used for agriculture, hence leaving some forest in the system; and (4) all the population stops the seasonal migration and becomes sedentary. In this last case, the abandonment of agriculture in the *inverneiras* causes complete forest re-vegetation while some forest can be maintained in the *brandas*.

An important result of the model is that the socio-ecological system is prone to regime shifts between the entire population (or the vast the majority) being sedentary and the entire population (or the vast the majority) being nomad. Recent transitions between the migrant lifestyle and the sedentary lifestyle in Castro Laboreiro support the model (Domingues & Rodrigues, 2008). As recent as 1985, the communities of 15 *brandas* still practiced nomadism to *inverneiras*, a practice that dated back at least since the XVIIIth century. By 2007, in 7 of those *brandas* the entire population had become sendentary, 4 were still inhabited by migrants, and another 4 had a mixed population of migrants and sedentary individuals. If the system did not exhibit regime shifts, one would expect that as some of the population became sedentary, most communities would exhibit a mixed composition of migrants and sedentary individuals, and this is not the case.

In our model people change their behavior according to the relative utilities of the nomadic and sedentary lifestyles. In Castro Laboreiro, better living conditions during the winter allowed people to remain in the summer villages (*brandas*) year round and, in comparison, made the migration more costly. This lead to the abandonment of many *inverneiras* and, according to our model, it should have lead to complete land abandonment and forest encroachment in those *inverneiras*. However a detailed analysis of land-use change patterns over the last 50 years shows a more complex dynamics (Rodrigues, 2010): the lower altitude and soil quality of the fields in the *inverneiras* are advantageous for the growth of crops and pastures, and increased mobility due to automobiles allow for maintenance of some fields even if people do not reside in the winter villages.

Other socio-ecological studies also illustrate the link between the decision making process of farmers and forest secondary successions (Perz & Stephen, 2002; Satake & Iwasa, 2006; Satake *et al.* , 2007). These models are consistent with the observed natural succession following the abandonment of agricultural land that occurred in several countries, leading to the regeneration of scrubland and woodland (Aide & Grau, 2004; Conti & Fagarazzi, 2005; Rey Benayas *et al.* , 2007). However, it is important to take into consideration the fact that forest regeneration is a long process that depends on the "cultivation legacy" (Cramer *et al.* , 2008) and can take more than 30 years (Verburg & Overmars, 2009). Further development of this socio-ecological model could take into account the intermediate stages of vegetation between agriculture and forest. It would also be interesting to consider how assuming constant rates of deforestation per resident, independently of the amount of forest available for harvest, would change the dynamics of the model presented here. It will probably increase the non-linear behavior of the system and the occurrence of regime shifts as the chances of complete forest clearing would likely increase.

More importantly the regime shifts found in the social-ecological model illustrate the importance of taking into account the interactions between the social and ecological components of ecosystems. Identifying the existence of tipping points in ecosystems is also necessary to assess their resilience and is particularly important when considering the ecosystem services provided (Carpenter *et al.* , 2009), especially when the resulting regime shifts are irreversible (Walker & Meyers, 2004; Leadley *et al.* , 2010; Rockström *et al.* , 2009).

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4. Maintaining disturbance-dependent habitats

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"Look deep into nature and you will understand everything better"

Albert Einstein.

Laetitia M. Navarro conducted the research and the redaction of the manuscript. Vania Proença contributed to the parts on fire dynamics, prescribed burning and diversity and intermediate disturbances. Jed O. Kaplan contributed to the part on the temporal evolution of the European landscape, particularly with the first figure of the chapter. Henrique M. Pereira contributed to developing and discussing the ideas presented in the chapter. All authors commented and discussed the manuscript.

Resumo:

Perturbações naturais (i.e. que não resultem de acção humana) são um processo essencial à dinâmica dos ecossistemas. Estas perturbações desempenham um papel importante, entre outros, na manutenção da estrutura dos ecossistemas e reciclagem dos nutrientes. Como tal, os regimes de perturbações, ou a sua inexistência, contribuíram para moldar a paisagem do nosso planeta e manter a sua diversidade de ecossistemas e de espécies que nele habitam. As dinâmicas de incêndios naturais e de herbivoria por megafauna selvagem, em particular, desempenharam um papel essencial na definição da paisagem da Europa antes do desenvolvimento da agricultura.

Estes fenómenos naturais foram com o tempo gradualmente substituídos por perturbações antropogénicas. O surgimento da agricultura e o desenvolvimento de sociedades complexas propulsionaram o declínio da megafauna Europeia, resultando em extinções locais, e até globais, de muitas espécies e alterações substanciais nos regimes de incêndios. Contudo, pela primeira vez desde a epidemia da Peste Negra, a exploração agrícola na Europa está a diminuir. Áreas marginais pouco produtivas têm sido progressivamente abandonadas ao mesmo tempo que as produções agrícolas e pecuárias se concentram nas zonas mais férteis e fáceis de cultivar. Com poucas ou nenhumas alternativas às perturbações antropogénicas associadas às práticas agrícolas abandonadas, há uma preocupação crescente com o possível desaparecimento de comunidades dependentes de perturbações, juntamente com os seus serviços de ecossistema.

Este capítulo pretende ilustrar as paisagens europeias ao longo do tempo, desde os povoamentos humanos pré-modernos ao progresso da agricultura e finalmente até à recente tendência de abandono agrícola. Também apresentamos dois regimes de perturbações principais: herbivoria por megafauna e incêndios naturais, de uma perspectiva histórica (i.e. perturbações não antropogénicas) e a sua importância para a preservação de certos tipos de habitats dentro da paisagem europeia.

Neste capítulo também investigamos a relação entre perturbações (tanto naturais como antropogénicas) e os níveis alfa, beta e gama de diversidade na paisagem. Estas perturbações não exercem apenas pressão selectiva na formação de comunidades, mas também reflectem a estrutura e composição dessas comunidades. Em particular, as alterações no uso dos terrenos provocadas pelo abandono rural podem criar as condições necessárias para um aumento dos eventos perturbadores, especialmente do risco de incêndios devido à acumulação de combustível e invasão por mato arbustivo, mas podem resultar também num menor número de perturbações se os seus agentes, tais como herbívoros domésticos de pasto, se tornarem residuais ou desaparecerem. Como resultado, uma ocorrência mais frequente de incêndios pode levar a uma homogeneização das comunidades e a um declínio em todos os níveis de biodiversidade. Nas zonas florestais, um menor regime de perturbações por pastoreio permitirá a expansão das florestas, igualmente diminuindo a heterogeneidade dos habitats e afectando espécies especializadas em pradaria.

Uma vez que o *rewilding* se apresenta como uma oportunidade para combater o problema do abandono agrícola, esta secção investiga opções para a preservação de habitats auto-suficientes e dependentes de perturbações. Em particular, sugerimos uma restauração assistida nas etapas iniciais do abandono, que pode ser feita através de incêndios controlados e o fomento ou introdução, quando necessária, de populações de mamíferos selvagens.

Finalmente, este capítulo lida com as dificuldades em escolher linhas de base para a conservação e sugere que linhas de base históricas devem servir apenas como referência e não como objectivos para a restauração.

Palavras-chave: Perturbações, regime de incêndios, habitats dependentes de perturbações, herbivoria, reintrodução, queimadas controladas

Maintaining disturbance-dependent habitats

Abstract: Natural disturbances, or the lack thereof, contributed to shape the Earth's landscapes and maintain its diversity of ecosystems. In particular, natural fire dynamics and herbivory by wild megafauna played an essential role in defining European landscapes in pre-agricultural times. The advent of agriculture and the development of complex societies exacerbated the decline of European megafauna, leading to local and global extinctions of many species, and substantial alterations of fire regimes. Those natural phenomena were over time gradually and steadily replaced by anthropogenic disturbances. Yet, for the first time since the Black Death epidemic, agricultural land-use is decreasing in Europe. Less productive marginal areas have been progressively abandoned as crop and livestock production has become concentrated on the most fertile and easier to cultivate land. With little or no substitute for the anthropogenic disturbances associated with these abandoned agricultural practices, there is growing concern that disturbance-dependent communities may disappear, along with their associated ecosystem services. Nonetheless, rewilding can give an opportunity to tackle the issue of farmland abandonment. This chapter first depicts the historical European landscapes and the role of two natural disturbances, herbivory and fire. The importance of disturbance-dependent habitats is then highlighted by drawing attention to the alpha and beta diversity that they sustain. Finally, the chapter investigates options for rewilding abandoned land to maintain disturbance-dependent and self-sustained habitats for which we suggest active restoration in the early stages of abandonment. This may be achieved via prescribed burning and support or introduction, when necessary, of populations of wild mammals.

Keywords: Disturbances, fire regime, disturbance-dependent habitats, herbivory, reintroduction, prescribed burning.

4.1. Introduction

Disturbance can be defined as "a discrete event that disrupts the structure of an ecosystem's community or population, and changes resources availability or the physical environment" (Turner, 1998). Natural disturbances (i.e., not deriving from human-induced processes) are an essential process of ecosystem dynamics. Among other roles, disturbances contribute to the maintenance of ecosystem structure and nutrient cycling (Attiwill, 1994; Turner, 1998). More important than considering the impact of a disturbance event per se is to consider the regime underlying disturbances and that characterizes the landscape (Turner, 1998), namely disturbance frequency and return interval, spatial extent, intensity (energy flow per area per time) and severity (magnitude of impact).

For millennia, humans have modified ecosystems with varying intensity and over various spatial extents. These anthropogenic changes imply a modification in both the natural communities and the natural processes that cause disturbance. In particular, human activities often cause the disruption of natural regimes, either directly (e.g., livestock grazing, fire suppression) or indirectly (e.g., landscape fragmentation, introduction of exotic invasives or pests), or introduce new types of disturbance, such as pollution. In addition, human activities can mimic natural disturbance regimes and affect biotic communities in a similar way (Attiwill, 1994). For example, the maintenance of traditional landscapes and the species-rich communities associated with them is implicitly linked with continuous ecosystem disturbance imposed by human activities.

If the regime of anthropogenic disturbances is altered, by a reduction or complete withdrawal of human activities, there is a concern that disturbance-dependent habitats and the associated communities may not be maintained. In particular, the maintenance of extensive farming systems in Europe is currently at stake due to farmland abandonment, which raises concerns about the potential effects of land-use changes on biodiversity (Rey Benayas *et al.* , 2007). The trajectory of ecological succession after abandonment depends on several factors, but the probable shift from a moderate disturbance regime (i.e. traditional landscape mosaic) to a low or high disturbance regime is associated with the risk of habitat homogenization and decline of species richness. Thus, one of the challenges of rewilding abandoned farmland is to contribute to the maintenance of disturbance-dependent habitats.

Passive regeneration following farmland abandonment can be a long and complex process, specific to each area (see chapter 2). It depends on the cultivation history, the time since abandonment, the availability of a "natural" seed bank, the proximity of sources of populations of species, and the requirements for natural disturbances, which will all take part in the self-sustained functioning of the restored ecosystem. When active restoration is needed, the choice of the baseline is also important (Corlett, 2012), and in this regard, open land maintained quasi exclusively by (traditional) agricultural practices is a rather recent norm.

In this chapter, we first depict the European landscapes through time, from pre-modern human settlement to the progressive advent of agriculture and finally to the recent trends of agricultural abandonment. We then present two major disturbances: i.e. herbivory and fire, from both a historical perspective (i.e. non-anthropogenic disturbance) and a restoration approach. We also look into the consequences of those disturbances on alpha and beta diversity levels in the landscape.

4.2. A picture of historical European landscapes

An ongoing debate...

Describing the species, habitats, and interactions that would be present without the influence of modern humans, i.e. the pre-historical baseline, is an important step to understand natural dynamics and disturbances, and guide the restoration of self-sustaining systems (Svenning, 2002; Gillson & Willis, 2004; Willis & Birks, 2006). However, the composition of the "pre-Neolithic landscape" (Hodder *et al.* , 2009) is still the subject of active debate.

The oceanic Middle and Late Pleistocene interglacial can be used as proxies to describe the European pre-Neolithic landscapes, due to their similar climatic conditions and low human activity (Svenning, 2002). Two contrasting pictures of lowland temperate European landscapes for these periods are described: (i) the "high-forest" hypothesis, where most of Europe was covered by forest, which dynamics and the resulting availability of open-land influenced herbivore populations; (ii) the "wood-pasture" hypothesis, depicting the European landscapes as a mosaic of forest and open-land where herbivory was the main driver of openness (Vera, 2000; Bradshaw *et al.* , 2003; Birks, 2005; Mitchell, 2005).

Pollen records are quite rich and repetitively used to test both hypotheses and assess the degree of openness, or lack thereof, of European landscapes. Typically, the ratio between the percentage of tree pollen and non-arboreal pollen gives an indication of the openness of a landscape (Svenning, 2002). Pollen records show that shade-intolerant species were present in areas both with and without evidence of large herbivores, which is in favor of the "high forest" hypothesis, in which grazers are not essential to maintaining those species (Mitchell, 2005). Nonetheless, pollen and dung beetle fossil record support the idea that megaherbivores were the main keepers of openness, at least of the floodplains in Northwestern Europe (Svenning, 2002), as a diverse community of dung-dependent beetles can be linked with the occurrences of large populations of herbivores (Sandom *et al.* , 2014).

Yet, three other types of natural processes can also explain the occurrence of open areas: forest fires, windthrows and edaphic-topographic conditions (Svenning, 2002; Fyfe, 2007; Molinari *et al.* , 2013). The most likely explanation is that the distribution of habitats was originally based on physical factors (Bradshaw *et al.* , 2003), and was then enhanced and/or maintained by large herbivores.

Temporal evolution of the European landscape

The first hominids reached Europe from Africa in the Early Pleistocene, some 1.2-1.1 million years ago (Carbonell *et al.* , 2008), while modern humans colonized the continent between 46 000 BP and 41 000 BP (Mellars, 2006). During this phase of migration, open areas were most likely more attractive for prehistoric human communities to settle in (Fyfe, 2007). The appropriation of new land coincided with changes in the European landscape. Nomadic hunter-gatherers started to actively manage their ecosystem with the use of fire during the Pleistocene: what started as a domestic tool (e.g. for cooking, heating, and for protection from predators) also became useful to draw game to hunting grounds, to clear travel routes, and to open space for grazers (Daniau *et al.* , 2010; Kaplan *et al.* , 2011; Pfeiffer *et al.* , 2013).

The development of agriculture was the next step in humans' appropriation and management of their environment (Pereira *et al.* , 2012 and see Appendix A). The spread of agriculture from the northern Levant and northern Mesopotamian area towards Europe has been calculated to have started between 11,550 and 9000 BP and expanded at a rate of 0.6 to 1.3 km/yr, with agriculture reaching north-western Europe in 3000 years (Pinhasi *et al.* , 2005; Ruddiman, 2013). Such spread of agriculture led to a fivefold increase in the human population (Gignoux *et al.* , 2011), which had considerable consequences on the landscape. Several models have been designed to investigate the historical evolution of this human impact. Typically, models that do not assume a direct linear link between human population density and deforestation, but also consider other factors, such as technological change, show that the rate of land appropriation was much higher in the distant past (Ruddiman, 2013; Kaplan *et al.* , 2011), which is supported by anthropological and archeological evidence (Farrell *et al.* , 2000). First of all, as time passed and deforestation occurred, less and less forest was left to clear. Most of all, technological improvements allowed people to produce the same amount of food on less land, which contradicts a direct link between population density and deforestation (Ruddiman, 2013). Following these non-linear concepts, Kaplan *et al.* (2011) presented model scenarios of Holocene anthropogenic land cover change. At 8000 BP, only Mesopotamia and Turkey were showing signs of human use of the land, but by the beginning of the Iron Age at 3000 BP, up to 40% of European land could have been cleared for extensive agriculture and pastures (Fig. 4.1). Between 8000 and 3000 BP, Kaplan *et al.* (2011) suggest that land use in Western Europe ranged from 5.5 to 6.5 ha per capita and was relatively stable. By 2500 BP, increasing populations in most of Western Europe triggered intensification of land use (Fig. 4.1) and decrease in per capita values. Major land abandonment episodes followed decreases in population during the Migration Period following the fall of the Western Roman Empire, and after the Black Death epidemic of AD 1350. By AD 1850, the latest preindustrial time, most of the European landscapes usable for intensive crop or pasture were deforested, and land use had dropped to values close to 0.5 ha per capita.

Following the Industrial Revolution, the relationship between population and land use had become largely uncoupled. Beginning in the late 18th Century, these “forest transitions” (e.g. Mather *et al.* , 1998) led to abandonment of unproductive agricultural and pastureland in most European countries. More recently, the rural population decreased by 17% since 1961 in Europe (FAOSTAT, 2010), with repercussions for agricultural land-use, and both are projected to continue decreasing in the decades to come. By 2030, up to 15% of the land cultivated in 2000 could be abandoned (e.g. Verburg & Overmars, 2009; Eickhout *et al.* , 2007) which represents 9.9 to 29.7 million ha of land (chapter 2). The areas facing the greatest likelihood of rural abandonment are remote and/or mountain areas, classified as “least favored”, with marginal value for agriculture (e.g. MacDonald *et al.* , 2000). With the withdrawal of human activities, those abandoned areas are often left without

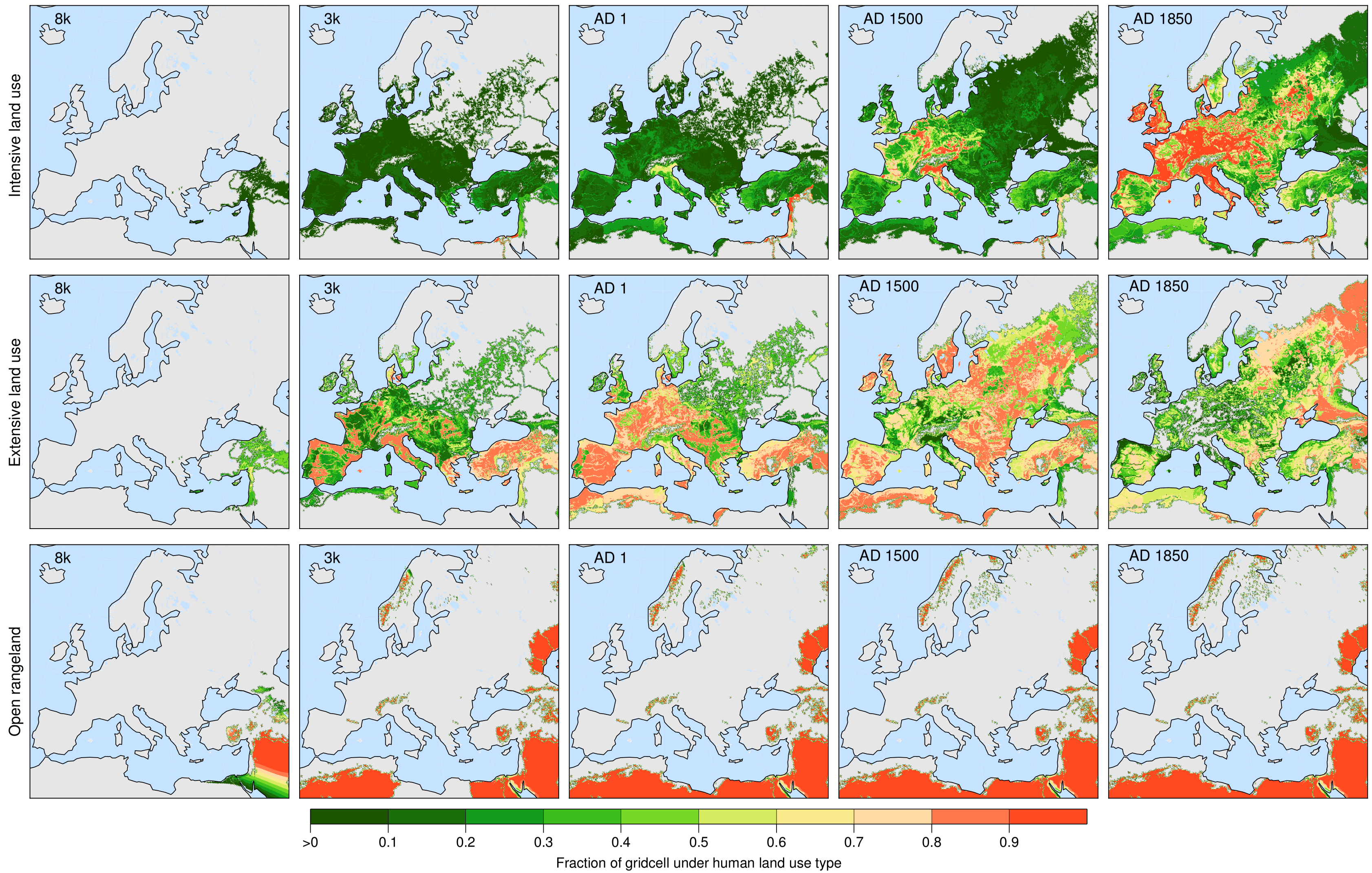

Figure 4.1.: Anthropogenic land cover change in Europe over the Holocene. Intensive land use includes land completely deforested and used for agriculture and pasture, while extensive land use includes forest-pasture, coppice and other managed forestlands. Open rangeland occurs on land that is either too cold or two dry for non-irrigated agricultural land uses. Land use is driven by estimates of past population at the country-level (Based on Kaplan *et al.* , 2011; Kaplan, 2012; Kaplan *et al.* , 2009).

the artificial disturbances that had replaced the natural ones, centuries or millennia ago.

4.3. The role of natural disturbances

Investigating the history of natural disturbances can inform researchers and managers on guidelines for restoration (Donlan *et al.* , 2006). Two types of disturbances were identified as fundamental in the maintenance of European landscapes, prior to human appropriation of the land: large herbivores and natural fire dynamics.

The pre-Neolithic ecosystem engineers

Ecosystem engineers are organisms that create and/or maintain habitats, either directly or indirectly (Jones *et al.* , 1994; Wright & Jones, 2006) and thus create niches for other species. The sole fact of grazing and browsing is not enough to be qualified as engineering (Wright & Jones, 2006), yet the consequences of herbivory, trampling, and fertilizing, especially by large herds of megafauna, have a direct impact on the distribution of habitats (Vera, 2000). Large herbivores can thus also be viewed as biotic drivers of land-cover change (Birks, 2005), or engineers. Small mammals are also known to have an important impacts on the vegetation, for example by disturbing the soil and modifying its physical and chemical properties (Jones *et al.* , 1994) but this goes beyond the scope of this chapter.

During the interglacial cycles of the late Quaternary, and prior to massive extinctions, the landscapes of Europe were characterized by a rich megafauna (Bradshaw *et al.* , 2003). The available fossil evidence can prove the presence of species in a given region, while using the impact of similar extant species as a proxy informs on the role of extinct megaherbivores on the landscape (Corlett, 2012). Nonetheless, in contrast with pollen, there are too little fossil records of pre-Neolithic large herbivores to allow for an estimate of their past densities (Bradshaw *et al.* , 2003; Mitchell, 2005), and we still lack precise knowledge regarding their behavior (Hodder *et al.* , 2009).

The late Pleistocene megafauna of Europe, until the early beginning of the Holocene (Tab. 4.1) resembles the one currently found in savannas, with herbivores such as proboscidae and rhinocerotidae, and large carnivores such as hyaenidae and felidae

(Vera, 2000; Bradshaw *et al.* , 2003; Blondel & Aronson, 1999). Globally, the group of large herbivores suffered more prehistoric extinctions than other taxa (Johnson, 2009). Cyclic climatic change had typically been responsible for a regular faunal turnover, and was later combined with increased human pressure (Corlett, 2012; Morrison *et al.* , 2007), leading to several of these megafauna becoming regionally (e.g. hippopotamus), globally (e.g. woolly mammoth), and often functionally, extinct (Bradshaw *et al.* , 2003; Blondel & Aronson, 1999). Some species also suffered large range contractions, such as the elk (Morrison *et al.* , 2007). Additionally, humans domesticated animals in the Fertile Crescent, about 10,000 years ago (Zeder, 2008; Pereira *et al.* , 2012), and as herders migrated west, increasing the area of pasture in Europe, wild herbivores were replaced by domesticated species. Since AD 1, most of the open rangeland in Europe has been under human land-use (Fig. 4.1).

Extinct and extant large herbivores can be classified according to their feeding behavior (Tab. 4.1), which in turn provides information on the type of habitat that their pressure can maintain(Vera, 2000; Svenning, 2002; Bullock, 2009): browsers (e.g. elk, straight-tusked elephants) are typically associated with tree rich areas; grazers (e.g. hippopotamus, aurochs) are in contrast associated with the occurrence of grass-rich habitat; finally, mixed feeders (e.g. red deer, wild goats) alternate between browsing and grazing. The European bison has, for example, been associated historically with both closed forest and semi-open habitats (Kuemmerle *et al.* , 2012). The social structure of the herbivores (i.e. solitary, groups or herds) also provides information on the grazing and browsing pressure on the landscape (Tab. 4.1).

The impact of large herbivores on their environment goes beyond browsing and grazing. For example, elephants are known to create large physical disturbances via the trampling of trees and shrubs (Jones *et al.* , 1994), which changes their habitat and the fire regime locally and in return benefits light demanding plant species. The disturbance induced by the rooting behavior of wild boars favors natural forest regeneration, while being considered as damaging to grasslands (Schley *et al.* , 2008; Sandom *et al.* , 2013b). Large herbivores also have a role as seed-dispersers through their consumption of large quantities of forage containing fruits, and which, due to low oral processing, disperse undamaged seeds in their feces (Corlett, 2012; Johnson, 2009). Some seeds even need to pass through a digestive track to trigger germination. Finally, herbivore dung is important for nutrient cycling and soil fertilization (Zimov, 2005).

Species	**Extant in the Holocene**	**Extinct**	**Extinct Europe**	**Extinct locally**	**European IUCN Status**	**Feeding behavior**	**Social structure**
Alces alces (Eurasian elk)	X			X	LC	B	S
Bison bonasus (European bison)	X			X	VU	M	H
Bison priscus (Steppe bison)		X				G	
Bison schoetensacki		X				G	
Bos primigenius (Auroch)	X	X				G	
Bubalus murrensis (Murr water buffalo)		X				M	
Capra aegagrus (Wild goat)	X				VU	M	H
Capra ibex (Alpine ibex)	X			X	LC	M	G
Capra pyrenaica (Iberian ibex)	X		X*	X	LC/VU	M	G
Capreolus capreolus (Roe deer)	X				LC	B	S
Capreolus suessenbornensis		X				M	
Castor fiber (Beaver)	X			X	LC	B	S
Cervus elaphus (Red deer)	X			X	LC	G	G
Coelodonta antiquitatis (Woolly rhinoceros)		X				B	
Dama clactoniana		X				M	
Dama dama (Fallow deer)	X				LC	M	H
Dicerorhinus hemitoechus (Narrow-nosed rhinoceros)		X				B	
Equus ferus (Tarpan)	X	X				G	
Equus germanicus (Forest horse)	X	X				G	
Equus hydruntinus (European ass)	X	X				G	
Equus przewalskii (Przewalski's horse)	X		X	X		G	
Hipparion crassum		X				B	
Hippopotamus amphibius (hippopotamus)	X		X	X		G	
Hippopotamus antiquus (European hippopotamus)		X				M	
Mammuthus primigenius (Woolly mammoth)	X	X				G	
Megaloceros cazioti		X				M	
Megaloceros dawkinsi (Giant deer)		X				M	
Megaloceros euryceros		X				M	
Megaloceros giganteus (Irish giant elk)	X	X				M	
Ovibos moschatus (Muskox)	X			X	LC	M	H
Ovis aries orientalis (Mouflon)	X			X	VU	M	G
Palaeoloxodon antiquus (Straight-tusked elephant)		X				M	
Pseudodama nestii		X				M	
Rangifer tarandus (Reindeer)	X			X	LC	M	H
Rupicapra pyrenaica (Pyrenean chamois)	X			X	LC	M	G
Rupicapra rupicapra (Chamois)	X			X	LC	M	H
Saiga tatarica (Saiga)	X		X		CR	M	H
Soergelia elisabethae		X				M	
Stephanorhinus kirchbergensis (Merck's rhinoceros)		X				B	
Sus scrofa (Wild boar)	X			X	LC	M	G
Ursus spelaeus (Cave bear)		X				M	

Table 4.1.: List of extant and extinct species of large herbivores present during the late Pleistocene and at some point between the early Holocene and the current time, in Europe (Bradshaw *et al.* , 2003; Bullock, 2009; Smith *et al.* , 2003; Svenning, 2002; Vera, 2000; Blondel & Aronson, 1999). IUCN Status: LC - Least Concern; VU - Vulnerable; CR - Critically endangered. Feeding behavior: M - Mixed feeders; G - Grazers; B - Browsers. Social structure: G - Groups; H - Herds; S - Solitary. (*) There are four subspecies of *Capra pyrenaica*, two of which are extinct - *C. pyrenaica lusitanica* and *C. pyrenaica pyrenaica* - while two are extant - *C. pyrenaica victoriae* and *C. pyrenaica hispanica*.

Fire dynamics

Fire is a critical component in the functioning of many ecosystems. It maintains and shapes vegetation structure and biotic communities, promotes natural regeneration and habitat diversity, takes part in biogeochemical cycles, and can influence soil properties and water functions (Bond & Keeley, 2005; Thonicke *et al.* , 2001). Unlike grazing, fires consume both dead and living material and do not discriminate between edible and non-edible plants (Bond & Keeley, 2005), but may act as a selective pressure over fire persistent traits(Pausas & Bradstock, 2007; Pausas *et al.* , 2006).

Fire-dependent systems cover about 53% of the world's terrestrial surface (Shlisky *et al.* , 2007). These systems evolved in the presence of fire and depend on this disturbance to maintain their structure and composition (e.g., Mediterranean forests and boreal forests), with fire regimes (characterized by their frequency, intensity, and seasonality) specific to each ecosystem. In addition, 22% of the world's terrestrial area is covered by fire-sensitive ecosystems, where fire plays a minor role in maintaining ecosystem structure and composition (e.g., broadleaved and mixed forests in the Alps), 15% is covered by fire-independent ecosystems, where fire is not an evolutionary force due to the scarcity of fuel or ignition sources (e.g. tundra), and the remaining 10% are not yet classified (Shlisky *et al.* , 2007).

In Europe, natural fire regimes are mainly of two types: i) intense and large, and ii) cool and small (Archibald *et al.* , 2013). The former type is typical of Mediterranean and boreal ecosystems, where large crown fires of high intensity return at intervals that can span from a few decades, in particular in Mediterranean regions, to more than a century (Archibald *et al.* , 2013). The latter type occurs interspersed with the first type, in the same biomes, and is associated with surface fires burning litter fuels (Archibald *et al.* , 2013). However, due to a long history of human presence, many ecosystems in Europe, including fire-sensitive systems, present altered fire-regimes resulting from land-use changes and anthropogenic fire management (Archibald *et al.* , 2013; Molinari *et al.* , 2013; Shlisky *et al.* , 2007). Current fire occurrence in Europe ranges from less than five to nearly a hundred per year in areas of the Mediterranean region, which also presents the largest average of area burned yearly, with over 10 000 ha/year in some areas (European Commission, 2010). Four types of areas can be identified in Europe, based on their fire regimes, when combining both the occurrences of fire and the average area burned (Fig. 4.2). Central France, North-Eastern Germany, and most of Romania present small fire regimes, with few fires (< 20 per year) and little area burned (< 35 ha). Poland, most of the Baltic and Scandinavian countries are areas with relatively high occurrences of fire (> 50 per year) but small area burned (< 35 ha). In contrast, most of Bulgaria and Greece are regions where a small number of fires (< 20 per year) are sufficient to burn large areas (> 115 ha). Finally, Southern Italy, Croatia and the Iberian Peninsula are areas with both high fire frequency (> 50 per year) and large areas burned (> 115 ha).

Fire suppression is a common land management policy implemented to protect human communities and land (Shlisky *et al.* , 2007; Fernandes, 2013) but it also promotes fuel accumulation in fire-dependent systems and increases the risk of large and intense fires (Fernandes, 2013; Proença & Pereira, 2010a). On the other hand, fire has also been extensively used as a tool to clear landscapes and reduce fire risk. In Europe, anthropogenic fires are often more frequent than natural fires. High frequency fire regimes can cause species community impoverishment, through the exclusion of fire sensitive species and the promotion of fire resilient species that can endure frequent fires, and it can also cause extensive soil degradation and nutrient loss (Thonicke *et al.* , 2001). This is particularly true for Mediterranean ecosystems, where 93% of fire regimes are considered to be in a degraded or very degraded state (Shlisky *et al.* , 2007).

Today, farmland abandonment is driving further changes in fire regimes across Europe, particularly in Southern Europe, with potential impacts for biodiversity and ecosystem services (Mouillot *et al.* , 2005; Bassi *et al.* , 2008; Proença & Pereira, 2010a). Where the number of ignitions is not a limiting factor, which is true in many regions under farmland abandonment (Bassi *et al.* , 2008; Ganteaume *et al.* , 2013), climate and fuel availability will be the main determinants of future changes to the fire regime. In high-productivity ecosystems with a high level of humidity, such as temperate broadleaved forests, fires will be limited by climate and humidity level, and less responsive to changes in fuel accumulation, since fuel is already a non-limiting factor (Pausas & Ribeiro, 2013). Vegetation will be more susceptible to fire during warmer seasons following droughts, when the existing fuel is more flammable (Pausas & Ribeiro, 2013; Proença & Pereira, 2010a). In low-productivity ecosystems, such as arid Mediterranean scrublands, fuel is the main limiting factor and will be the main driver of shifts in the fire regime (Pausas & Ribeiro, 2013; Pausas & Fernández-Muñoz, 2012).

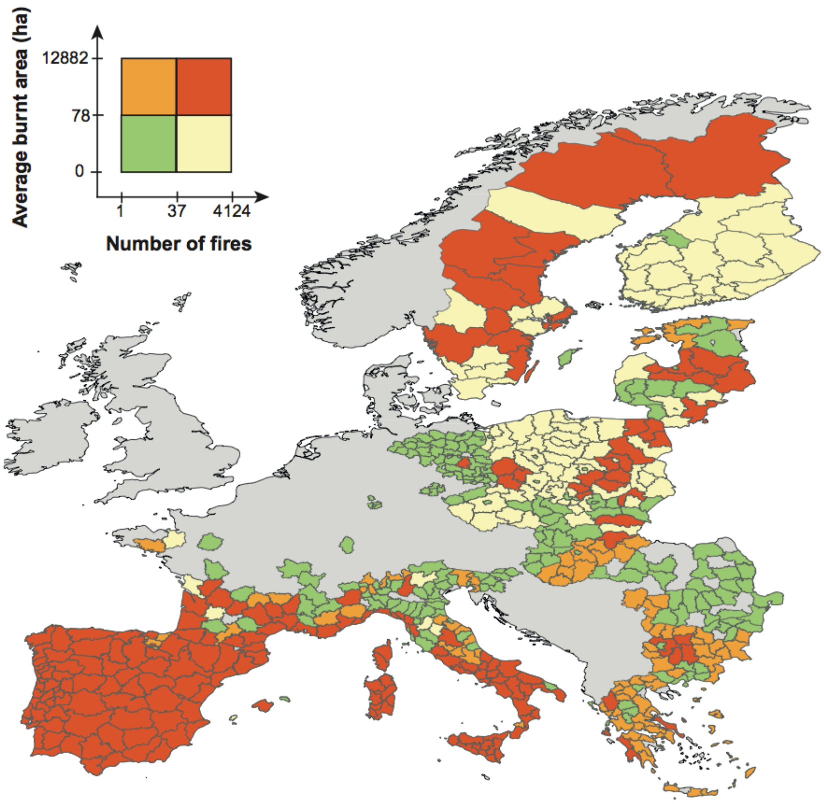

Figure 4.2.: Occurrence and intensity of fires in Europe over the 2005-2010 period. The average yearly occurrence of fire and average area burned (ha) for the 2005-2010 period, per NUTS3 administrative unit were calculated, only including NUTS3 for which data were available for at least 4 years. The double color ramp allows to identify areas with high number of fire but low area burned (yellow), areas with low occurrence of fire but large burned areas (orange), areas with few fire and small areas burned (green), and areas with both high occurrence of fire and large burned areas (red). Source: EFFIS for the fire data (European Commission, 2010) and © EuroGeographics for the map of administrative boundaries.

Recent trends in the Western Mediterranean Basin support the above predictions (Pausas & Fernández-Muñoz, 2012). In this region, fields used to be grazed, frequently burned (small scale) and cleared for farming and timber (Proença & Pereira, 2010a), limiting fuel availability. The rural exodus since the mid-20th century led to shrub encroachment and afforestation with fire-prone species, and resulted in more frequent, more intense and larger fires. Today, increased fuel load and spatial continuity are driving a shift in the fire regime, which is becoming more responsive to drought, similar to high-productivity ecosystems (Pausas & Fernández-Muñoz, 2012). In the future, the response of fire regime to changes in climatic variables, such as precipitation, is expected to be non-linear (Batllori *et al.* , 2013): while a small decrease in annual precipitation may increase probability of fire, a large decrease may lead to the inverse response due to a drop in ecosystem productivity, leading the system back to a fuel-limited fire regime.

4.4. Disturbances and diversity

Traditional landscapes in Europe, in particular High Nature Value (HNV) farmland areas, are acknowledged for their high species richness and conservation value (MacDonald *et al.* , 2000; EEA, 2004; Blondel & Aronson, 1999). The concept of HNV farmland includes three main types of managed landscape (EEA, 2004): farmland with a high proportion of semi-natural vegetation; farmland dominated by low intensity agriculture or a mosaic of semi-natural and cultivated land and small-scale features; and farmland supporting rare species or that concentrates a high proportion of populations of European or world distributed species. The first two types are particularly associated to extensive farming systems, and the third type acknowledges the value of some locally intensive farming systems for biodiversity conservation (Paracchini *et al.* , 2008). These types are not mutually exclusive and a particular landscape can encompass more than one type (Paracchini *et al.* , 2008). For example, extensive farmland landscapes are characterized by fine-grained habitat mosaics and/or by continuous and moderate disturbance arising from human activities and landscape management.

Species diversity patterns in traditional landscapes are likely to be different from what would be found in non-modified (primary) landscapes (Blondel & Aronson, 1999). The richness of species at the habitat patch scale (i.e., α-diversity) is probably lower in the case of specialist species in traditional mosaics due to the effect of habitat fragmentation and their low tolerance to the conditions found in other habitats (Proença & Pereira, 2013). For instance, the diversity of forest species is lower in fragmented forest patches than in an area of similar size in continuous habitat (Proença, 2009). On the other hand, if the total number of species is considered, then a higher α-diversity is expected in traditional landscapes due to species being able to use more than one habitat (except strict habitat specialists) and due to the high density of habitat edges, which facilitates inter-patch movements and therefore leads to a higher species turnover in space and time (Proença & Pereira, 2013; Guilherme & Pereira, 2013). Note that even with inter-patch movements, each habitat type will support a distinct community of species due to differences in species abundances and due to the existence of strict habitat specialists. Regarding species turnover (i.e., β-diversity), traditional landscapes can have a higher turnover than former undisturbed land (Blondel & Aronson, 1999), due to their mosaic structure. However, the soundness of this assumption depends on the scale of analysis (see Merckx, in press). For example, one can predict that the replication of the traditional habitat mosaic across large spatial scales results in a higher similarity of (modified) habitats, which promotes the presence of similar communities across large areas. Finally, the effect of these changes on the total number of species found in the landscape (i.e., γ-diversity) is less straightforward. Indeed, whilst several species suffered declines or even extinctions due to habitat destruction or modification (e.g., bear, auroch), other species benefited from these changes and proliferated in the human-modified habitats (e.g., farmland birds). Moreover, starting in the earliest Neolithic, farmers continually and intentionally introduced new species to European ecosystems (Blondel & Aronson, 1999). They also did so unintentionally as a result of species dispersal by animal herds along transhumance routes (e.g. Poschlod *et al.* , 1998). Both of these activities thus increased the regional species pool.

Diversity and intermediate disturbance

The intermediate disturbance hypothesis (Connell, 1978) and the diversity-disturbance hypothesis (Huston, 1979) are often used to explain the ecological mechanisms determining the high diversity of species found in traditional landscapes (e.g. Blondel, 2006): species diversity peaks when communities are exposed to moderate disturbance, in terms of frequency, extent and intensity. This occurs because moderate

Figure 4.3.: Minifundia system in a mountain landscape in Northwestern Portugal (Photo credit: Vânia Proença).

disturbance (e.g., moderate grazing) creates discontinuities in the ecosystem that allow the maintenance of early successional species while preventing dominance of more competitive species, hence keeping the ecosystem in a transitive state between early and steady-state communities. The management of traditional landscape mosaics (Fig. 4.3), with low intensity farming, moderate grazing and maintenance of forest patches, is often described as an example of intermediate disturbance, and therefore as a promoter of species diversity (Ostermann, 1998; Henle *et al.* , 2008). Nonetheless, peaked relationships between species richness and disturbance are not the rule across ecology studies (Mackey & Currie, 2001). Peaked curves are more commonly reported by studies covering small spatial scales and in the presence of natural disturbance regimes (Mackey & Currie, 2001). In addition, the relationship between taxa richness and the intensity of anthropogenic disturbance regimes is often non-significant (Mackey & Currie, 2001), increasing the challenge of predicting the impacts of altered regimes of disturbances on biodiversity.

Effects of land-use change on disturbance regimes

Land-use changes caused by rural abandonment can create the conditions for an increase in disturbance events, in particular higher fire risk due to fuel accumulation and shrub encroachment, but may also result in fewer disturbances if disturbance agents, such as domestic grazers or browsers, become residual or even disappear. The trajectory of secondary succession after abandonment depends on several interacting factors and ecological filters, such as the pool of colonizer species in the surrounding landscape, their ability to colonize abandoned patches, soil quality, and, of course, disturbance regime (Cramer, 2007). Disturbances will not only exert a selective pressure on community assembly, but will also respond to community structure and composition.

In landscapes where tree density is very low, such as some Mediterranean landscapes, there is a high probability of shrub encroachment after farmland abandonment due to seed limitation, predatory pressure over oak acorns and deficient abiotic conditions, such as poor soils (Acácio *et al.* , 2007). Wildfire will further promote shrub dominance, due to many shrubs' resprouting ability. Wildfires may hence establish a reinforcing feedback loop, leading to community homogenization and a decline in diversity at all scales (Proença & Pereira, 2010a).

A different trajectory can be anticipated in landscapes with a higher tree density, such as semi-natural grasslands in northern Europe (Eriksson *et al.* , 2002). There, seed availability and dispersal are not limiting factors and forest is able to colonize and regenerate in relatively short time. With an expected low disturbance regime, forest can expand, which declines habitat heterogeneity. Some species, such as grassland specialists, will show strong reductions in abundance or even go locally extinct. Impacts at the landscape level will depend on species ability to persist in alternative habitats such as forest edges or heathlands (Proença & Pereira, 2013).

The above examples describe abandoned patches in a fairly homogenous landscape matrix with either a low or high tree density. In a heterogeneous landscape with a more balanced cover of different habitats and a variety of edaphic-topographic conditions, scenarios would probably be different given the diversity of local responses to changes in disturbance regime. Habitat diversity will not only counteract landscape homogenization, but also provide alternative habitats for species affected by farmland abandonment, thus reducing the impact of land-use change on species diversity. The persistence of those species in the landscape will then depend on the maintenance of those alternative habitats, either by natural processes, such as herbivory by wild ungulates, or through assisted processes, such as prescribed fire or herbivore re-introduction.

4.5. Maintaining disturbance-dependent habitats

Wild herbivores: natural (re)colonization or (re)introduction?

Today, only 16% of the Palearctic region, including Europe, contains areas occupied by undisturbed large mammal faunas, i.e., species that have not undergone major changes in range between AD 1500 and the present (Morrison *et al.* , 2007). This figure does not even consider the number of species that went extinct early in the Holocene (Tab. 4.1). There is also a clear regional difference when looking at the current species richness of large herbivores in Europe (Fig. 4.4): countries of central Europe present the highest diversity, while the Westernmost countries have low richness, best explained by past local extinctions. Species rich areas, with lower human densities and less pressure on the land, could become “sources” for the natural re-colonization. This has already been documented for some species of large herbivores that show substantial increases in their populations since the 1960s (Tab. 4.2). Though legislation and conservation measures largely contributed to it (Deinet *et al.* , 2013), rural depopulation and the associated reduced human pressure, both direct (e.g. less hunting) and indirect (e.g. more land available), can also explain the phenomena (Tab. 4.2). Wild populations can also benefit from the absence of competitor and predator species (Bradshaw *et al.* , 2003), though unregulated population growth can quickly become an issue, e.g. if their pressure on the land is too high.

In cases where no local wild population is extant, as for example in Western European countries (Fig. 4.4), herbivores can be introduced, to restore ecosystem functioning (Sandom *et al.* , 2013a). That is, provided that their functional role is left unattended Lipsey & Child (2007), and that the abandoned land meets their requirement in natural resources. A study on fenced populations of wild boar showed that their rooting behavior can create germination niches (Sandom *et al.* , 2013b) and contribute to forest regeneration. However, they can also be detrimental to the established trees when bark stripping and uprooting (Sandom *et al.* , 2013a). Rein-

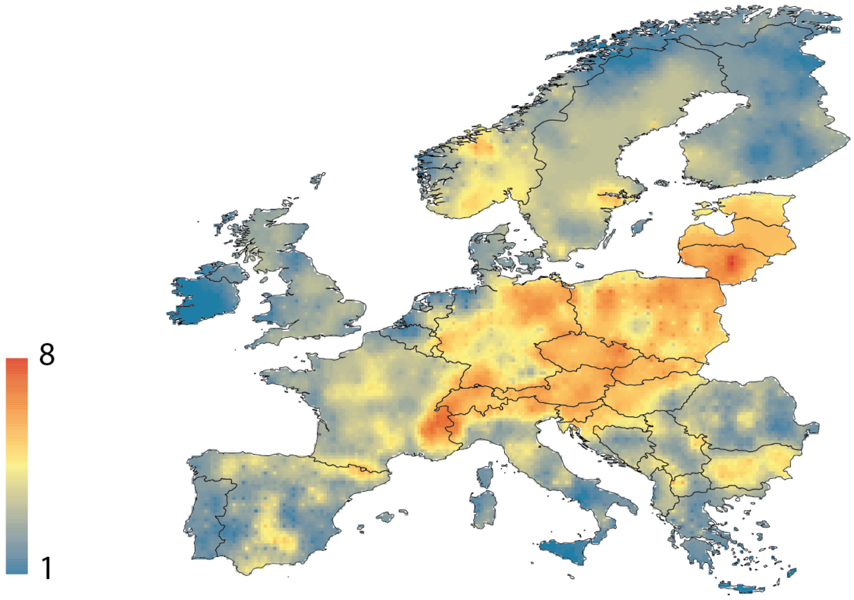

Figure 4.4.: Species richness for extant large herbivores of Europe - See Tab. 4.1 for the list of species. Map obtained using Inverse Distance Weighting (weight = 2) on the atlas data. Source: Atlas of European Mammals (Mitchell-Jones *et al.* , 1999).

troducing ecosystem engineers to restore and/or maintain disturbance dependent habitats can also be more time and cost effective than man-made restoration (Sandom *et al.* , 2013b; Byers *et al.* , 2006). Moreover, provided that the re-introduced species present charismatic values, their presence could facilitate the acceptation of a rewilding project by the public (Lipsey & Child, 2007; Kuemmerle *et al.* , 2012). The reintroduction of wild grazers can also be assessed positively from the standpoint of ecosystem services, based on the existence value of the megafauna (Proença *et al.* , 2008) and associated cultural services (see chapter 5). Nonetheless, a balance must be maintained when considering the (re)introduction of herbivores and many potential challenges should be raised and discussed (Seddon *et al.* , 2007; Corlett, 2012; IUCN, 2013a). First, which species should be reintroduced? When taxon substitutions are needed for ecological replacement (IUCN, 2013a), researchers' opinions are divided, ranging from releasing breeds of domesticated animals, to the reintroduction of extant parents of long gone species (e.g. Donlan *et al.* , 2006). Releasing animals also raises the question of increasing the risk of conflicts between local human populations and "wildlife" (e.g. Enserink & Vogel, 2006; Goulding & Roper, 2002), which could be more easily accepted if the species was progressively, and naturally, recolonizing an area. Those conflicts are linked to the liability of the organization that performed the reintroduction, (e.g. in cases of damages or accidents), and a legal framework for the introduction of domestic species (e.g. horses) is missing. Thus, one should question whether the reintroduced population should be free roaming or fenced. Finally, an overabundance of certain species can have detrimental effects, especially when the natural predator guild is absent and cannot regulate the populations (e.g. Boitani & Linnell, in press), yet no specific guidelines are designed for the natural control of reintroduced populations (IUCN, 2013a). For instance, the large populations of browsers in the Scottish Highlands, where large carnivores have been extinct for centuries, currently limit the natural forest regeneration (Sandom *et al.* , 2013a).

The choice between natural recolonizations, reinforcement of local populations or reintroductions will depend on the current distribution and abundances of the herbivore communities. In areas of Central Europe, one might expect that the diversity of herbivores is high enough to allow for recolonizations, while in Western and Southern Europe, active introduction might be needed (Fig. 4.4). In all cases, conservation measures, legislation and reduced human pressure are necessary for the establishment of viable populations (Tab. 4.2).

Prescribed burning

Fire can be used as a tool in landscape management for two main intents: to control fire risk and the intensity of wildfires, and to manage landscape structure and biodiversity. Prescribed fires are often used as a preventive measure to control fuel load and fire intensity (e.g. Fernandes, 2013). In addition, the combination of different fire regimes can be used to maintain landscape heterogeneity and habitat for species dependent on different ecosystem successional stages (Driscoll *et al.* , 2010). In regions where fire risk and shrub encroachment are paired threats to biodiversity conservation, fire can be used as a tool to approach both problems (Moreira & Russo, 2007). Nonetheless, the use of prescribed fires can also raise some conservation issues. For instance, prescribed fires are performed during the wet season (winter to spring) when there is a low risk of fire spreading, while natural fires occur during dry days, especially in summer. This divergence in fire season can negatively impact species that reproduce in spring (van Andel & Aronson, 2012), such as ground nesting birds, but also the persistence of plant species, for example by causing pre-

Species	Pop° estimate (year)[1]	Population increase (1960-2005)	Ranked causes for increase[2]: Legislation	Conservation measures (including reintroductions)	Reduced human pressure (e.g. land abandonment, decreased hunting)	Change in environmental conditions	Natural recolonization?
Alces alces (Elk)	719 810 (2004/2005)	+ 210%	3	1	2	4	Y
Bison bonasus (European bison)	2 759 (2011)	+ 3 000%		1		2	
Capra ibex (Alpine ibex)	36 780 (2004/2005)	+ 475%	1	2		3	
Capra pyrenaica (Iberian ibex)	> 50 000 (2002)	+ 875%	2	1	3		
Capreolus capreolus (Roe deer)	9 860 049 (2005)	+ 240%	3	2	1	4	Y
Castor fiber (Beaver)	> 337 539 (2003-2012)	+ 14 000%	1	2	3		Y
Cervus elaphus (Red deer)	2 443 035 (2002-2010)	+ 400%	2	1	3	4	Y
Rupicapra pyrenaica (Pyrenean chamois)	69 100 (2008)	+ 550%	1	2	3		
Rupicapra rupicapra (Chamois)	485 580 (2004/2005)	+ 85%	4	1	2	3	
Sus scrofa (Wild boar)	3 994 133 (2004-2012)	+ 400%		1	3	2	

Table 4.2.: Population trends for large herbivores in Europe and main reasons for recovery of the populations (based on Deinet *et al.* , 2013). (1) Some population estimates are obtained by summing values of national assessments performed in different years, hence the time intervals instead of the year of assessment in some cases. (2) Ranking based on Deinet *et al.* (2013), with "1" being the most relevant and "4" the least relevant. The number of observed causes of increase varies from species to species.

mature seed release, or by destroying seedlings of annual plants before they create a seed bank (Whelan, 1995; Bowman *et al.* , 2013). Another issue is the implications of prescribed fires for climate change mitigation. Large scale prescribed fires may aggravate climate change, due to the emission of greenhouse gases and aerosol particles (Russell-Smith *et al.* , 2009; Fernandes, 2013). While more research is needed to understand the effects of prescribed burning on carbon cycle (Fernandes, 2013), it is also accepted that well planned prescribed burning prevents larger losses of carbon to the atmosphere by reducing wildfire risk (Bowman *et al.* , 2013; Fernandes, 2013). Finally, defining the regime of prescribed fires can be challenging (Whelan, 1995; van Andel & Aronson, 2012). Replicating natural fire regimes may not be possible, due to the lack of historic information. It may even not be advisable, given changes in landscape structure and, in some areas, in local climate, which may lead to unpredicted responses to fire (Driscoll *et al.* , 2010). Therefore induced fire regimes should be planned to meet the desired outcomes instead of trying to mimic the parameters of natural fire regimes (Whelan, 1995).

4.6. Concluding remarks

Millennia of human activities have progressively replaced natural disturbances, such as herbivory and fire, to shape the European landscapes. Maintaining disturbance-dependent habitats after the withdrawal of those human activities is a difficult restoration process. It can be guided by knowledge of the past (Vera, 2000; Gillson & Willis, 2004; Willis & Birks, 2006; Sandom *et al.* , 2013a), and by improving our ability to understand ecosystem dynamics and projecting potential restoration pathways. This means identifying the most desirable outcome in terms of both biodiversity and resilience. Nonetheless, besides human impacts on the landscapes, other biotic and abiotic alterations have also led to the current ecosystem composition. The climate has changed during the past millennium and some species have gone extinct while others have invaded, all these changes influencing ecological processes (Gillson & Willis, 2004; Hodder *et al.* , 2009). The interaction between human pressure and natural changes (e.g. non-anthropogenic climatic changes) could also have led to the crossing of tipping points (Gillson & Willis, 2004; Kaplan *et al.* , 2011). Returning the landscapes to their historical conditions would thus be unachievable, if ever even desirable. This means that the baseline must shift, not only for the policy makers and the public who attribute cultural values to a relatively recent landscape (Vera, 2009), but also for scientists and conservationist, some of which, on the contrary, having too long of a memory of the European landscape.

An additional concern emerges with farmland abandonment, when, with a decrease in agricultural activities, herbivores also become functionally extinct (Donlan *et al.* , 2006; Pereira *et al.* , 2012), while the artificial fire regime is altered. Hence, in the early stages after land abandonment, the “restoration goals” must be defined to determine the set of biotic and abiotic factors that might be managed (Byers *et al.* , 2006). Supporting local populations of wild herbivores, reintroducing them in places where they are absent and using prescribed burning can constitute the first steps towards restoring self-sustained ecosystems, though all those management methods have their limitations which urge for more research on their possible applications. Moreover, both herbivores and fire regimes must be regulated as well (e.g. by predation for the former) to allow for self-sustaining ecosystems.

Though rewilding, in practice, is still debated, some large-scale initiatives have already been implemented, mainly to restore disturbance-dependent habitats and assess the role of re-introduced wild grazers. Listing all those initiatives goes beyond the scope of this chapter (but see Helmer *et al.* , in press) but the management approaches of some of those projects exemplify the need for a consensus on the concept of rewilding itself. Probably the most famous example is the Oostvaardersplassen in the Netherlands, a large scale (6000 ha), 40 year long project on the reintroduction of wild herbivores (Sandom *et al.* , 2013a; Sutherland, 2002). However, despite some successes, including the natural recolonization of some previously extinct bird species (Vera, 2009), the degree of rewilding is extremely controlled in the area since it is fenced, animals are sometimes fed in the winter, and the natural top-predator’s guild is absent. When rewilding is meant to lead to ecological restoration, reintroductions should be one of the tools rather than a goal per se. Moreover, historical baselines should be treated as guidelines, not as objectives. In other words, rather than focusing on the conservation of a given set of species or habitats, rewilding, which is a process-based conservation measure after all (Byers *et al.* , 2006), must focus on the restoration and preservation of ultimately self-sustaining ecosystems.

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5. Ecosystem services: opportunities of rewilding in Europe

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"The wilderness holds answers to questions man has not yet learned to ask."

Nancy Newhall

Yvonne Cerqueira conducted the redaction and Laetitia M. Navarro contributed to writing sections of the chapter. L.M.N. and Y.C. performed the qualitative mapping analysis and interpreted the results. L.M.N. performed the quantitative analysis. All the co-authors participated in developing the ideas and commented on the manuscript.

Resumo:

Travar a degradação e restaurar a plena capacidade dos ecossistemas de fornecer serviços é actualmente um importante compromisso político na Europa. Apesar de ser ainda um tema sob debate, o actual abandono agrícola na Europa é visto como uma oportunidade para que uma nova visão económica e de conservação ecológica tenha lugar, com a restauração dos processos naturais através do retorno ao estado selvagem, ou *rewilding*, como opção de gestão ambiental. No entanto, apesar do interesse ecológico em restaurar uma Europa mais selvagem, é necessário encontrar argumentos baseados em evidências e explorar os impactos de amplo alcance do *rewilding*.

Neste capítulo analisamos primeiro os padrões espaciais dos serviços de ecossistema tanto na UE25 como em zonas selvagens. Através desta análise, identificamos áreas com elevada oferta de serviços de ecossistemas, as quais coincidem principalmente com áreas (semi) naturais e de floresta, em particular nas regiões montanhosas. Discutimos também o estado e as ameaças, resultantes de alterações no uso do solo, ao fornecimento dos serviços de ecossistema.

Em seguida, procedemos com uma análise qualitativa da oferta de serviços de ecossistema nas regiões da Europa em que uma elevada qualidade de regiões selvagens (*wilderness*) foi encontrada. Em particular, identificamos quatro principais tipos de regiões: baixa oferta de serviços de ecossistemas e baixa qualidade dos habitats selvagens; baixa oferta de serviços de ecossistemas e elevada qualidade dos habitats selvagens; elevada oferta de serviços de ecossistemas e baixa qualidade dos habitats selvagens; e elevada oferta de serviços de ecossistema e elevada qualidade dos habitats selvagens. Em primeiro lugar, determinamos que algumas das áreas de elevada qualidade de habitats selvagens encontram-se associadas às regiões, principalmente em sistemas montanhosos, que fornecem mais serviços de ecossistema. A distribuição espacial dos serviços de regulação coincide também relativamente bem com a qualidade destes habitats selvagens, em que grandes áreas da Europa

contêm tanto elevada oferta de serviços de ecossistema como elevada qualidade de habitats selvagens (ex. Norte da Peninsula Ibérica e Áustria). Finalmente, para os serviços recreativos, encontramos uma predominância de áreas tanto de baixa oferta de serviços e baixa qualidade selvagem como elevada oferta de serviços e elevada qualidade dos habitats selvagens.

Seguidamente, exploramos quantitativamente a oferta de serviços de ecossistema na extensão da Península Ibérica. Através da comparação da distribuição dos valores de indicadores para a oferta de serviços de ecossistema entre “áreas actualmente cultivadas”, “áreas cultivadas, mas com abandono previsto” e “áreas nos 5% melhores habitats selvagens ibéricos”, podemos avaliar se um aumento ou diminuição na oferta dos serviços de ecossistema estudados ocorreria se o abandono agrícola e o *rewilding* ocorressem. Em particular, colocamos a hipótese de que o fornecimento de água doce, as reservas de carbono, a regulação do ciclo da água, a qualidade do ar e as áreas recreativas aumentariam se a área de habitats selvagens fosse expandida na Europa. Como resultado, e não surpreendentemente, a produção de alimentos também diminuiria nestas áreas. Discutimos ainda o potencial do *rewilding* para um aumento da oferta tanto dos serviços de regulação como culturais, investigando casos documentados de maior oferta de serviços de ecossistema associados ao *rewilding* e/ou a áreas selvagens.

Assumindo que habitats selvagens de elevada qualidade são uma boa aproximação ao *rewilding*, os nossos resultados sugerem que esforços para o retorno dos habitats selvagens em toda a Europa iriam reforçar a capacidade dos ecossistemas em fornecer elevados serviços culturais e de regulação, argumentando assim a favor do *rewilding* como uma ferramenta de restauração de terras agrícolas abandonadas.

Palavras-chave: serviços de ecossistemas, habitats selvagens, benefícios, abandono agrícola, bem-estar humano, retorno de habitats selvagens (*rewildimg*)

Ecosystem services: opportunities of rewilding in Europe

Abstract: Halting the degradation and restoring the full capacity of ecosystems to deliver ecosystem services is currently a major political commitment in Europe. Although still a debated topic, Europe´s on-going farmland abandonment is seen as an opportunity to launch a new conservation and economic vision, through the restoration of natural processes via rewilding as a land management option. Despite the ecological interest of restoring a wilder Europe, there is a need to generate evidence-based arguments and explore the broad-range impacts of rewilding. In this chapter we contribute to the on-going debate on rewilding by first analyzing the spatial patterns of ecosystem services in both the EU25 and in wilderness areas. We subsequently quantitatively explore the supply of ecosystem services in the top 5% wilderness areas, on agricultural land, and on land projected to be abandoned, at the extent of the Iberian Peninsula. We determine that high quality wilderness is often associated to high supply of ecosystem services, mainly regulating and cultural, specifically in mountain regions. Assuming that high quality wilderness is a good proxy for rewilding, our results suggest that rewilding efforts throughout Europe will enhance the capacity of ecosystems to supply high regulating and cultural ecosystem services, such as carbon sequestration and recreation.

Keywords: ecosystem services, wilderness, benefits, farmland abandonment, human well-being, rewilding

5.1. Introduction

Ecosystem services have been defined as the benefits humans derive from nature through a set of ecosystem functions. The Millennium Ecosystem Assessment (MA) was the stepping stone in providing a conceptual framework for ecosystem services as well as, assessing the consequences of ecosystem change for human well-being and the overall global trends. Since its publication (MA, 2005), multiple classification schemes for ecosystem services have been proposed, such as The Economics of Ecosystem Biodiversity (TEEB, 2012) and, more recently, the CICES (Common International Classification of Ecosystem Services). Adopted by the European Commission for the new Biodiversity Strategy for 2011-2020 (Maes *et al.* , 2013), the CICES categorizes ecosystem services into 3 groups (Haines-Young & Potschin, 2012): provisioning (e.g. food, fiber, fuel and water), regulating and maintenance (e.g. air quality, water and soil regulation, natural hazard regulation, climate regulation and disease control) and cultural (e.g. recreation and spiritual).

Although society can easily perceive provisioning ecosystem services such as crops, fish and freshwater, which are all direct benefits to humans, others, such as pollination, erosion control and climate regulation are less tangible. However, directly or indirectly, all ecosystem services underpin environmental and human well-being, economy, and businesses (MA, 2005). Many services are not traded in the conventional markets and hence, their economic values remain invisible, tending to be undervalued and consequently overexploited (de Groot *et al.* , 2012). Yet, once lost, replacement can be costly. Wetlands, for example, provide numerous regulating services (e.g. water purification and flood/storm protection), which are unnoticed, in contrast to provisioning services (e.g. timber and food), but highly valuable since degradation can lead to high replacement costs (Reed *et al.* , 2013).

Throughout the world, ecosystem services have been used as a tool in conservation and development as well as poverty alleviation (Tallis *et al.* , 2008). However, many conservation efforts have been unsuccessful due to human mismanagement of ecosystem services. The awareness that ecosystem services affect human well-being and economic development has resulted in their integration in policies and government strategies and the most recent EU Biodiversity Strategy European Commission (2011b). This new strategy sets goals to halt both biodiversity loss and the degradation of ecosystem services. In particular, it includes the protection of wilderness, specifically old growth forest. Today, 45% of Europe's land cover is forest (1 billion ha) but only 4% is undisturbed forest (6 million ha). Protecting these ecosystems is important as they support high quality ecosystem services, such as recreation and air quality (Maes *et al.* , 2012b). Increasing these percentages could greatly improve the supply of ecosystem services provided by natural habitats of no human influence. This new conservation strategy goes hand in hand with a fairly recent initiative, “Rewilding Europe” which aims to rewild one million hectares of land by 2020 (Helmer *et al.* , in press). In particular, the emergence of the rewilding topic is seen as an opportunity for rural areas, which have undergone land abandonment throughout the past decades (see chapter 2). However, we have yet to determine if rewilded areas will promote the supply of ecosystem services fundamental for human well-being.

In this chapter, we first investigate the supply and spatial distribution of ecosystem services on a pan-European scale before comparing it with the occurrence of wilderness areas. We then use the supply of ecosystem services in wilderness areas as a proxy for areas that are projected to undergo land abandonment and rewilding. Finally, we discuss the various economic and ecological benefits of rewilding in Europe.

5.2. Europe and Ecosystem services

Current supply of ecosystem services

Ecosystems provide a number of essential services underpinning all human life and activities. However, the continuance of the various ecosystem services is only possible through the recognition of ecosystems’ multiple functions integrated in management strategies. To manage for multiple ecosystem services we need to map and identify the spatial synergies and trade-offs between services (Raudsepp-Hearne *et al.* , 2010; Maes *et al.* , 2012b). In doing so, we are able to identify ecosystems supporting high level of services and biodiversity (Chan *et al.* , 2006). Along the years, the number of studies mapping ecosystem services and scenario building has grown, informing both planners and decision makers, prioritizing the protections and management of ecosystems (Chan *et al.* , 2006) and additionally, delineating cost-effective measures (Egoh *et al.* , 2008; Naidoo & Ricketts, 2006).

The integration of ecosystem services into current conservation strategies ensures future sustainability. However, this integration has only recently scratched the surface. In the EU Biodiversity Strategy for 2020, the need for spatial assessment of ecosystem services has been included as one of the key actions (European Commission, 2011a). Under Action 5, all EU Member States are required to map and

Service	Indicator	Unit	Description / Benefit
Food provision	HANPP	gC/m2/yr	Human Appropriation of Net Primary Production (cropland and grassland in this study).
timber provision	total stock of timber	m^3/ha	Production for fuel, construction and paper. Forest connectivity.
freshwater provision	Surface Water Flow (QFS)	mm	Renewable freshwater provision.
Climate regulation	Carbon stock	ton/ha	Above- and below-ground carbon stored in living plant material.
	Net Ecosystem Productivity (NEP)	mg/m^2/year	Carbon sequestration.
Water regulation	Nitrogen retention	%	Capacity of ecosystems to retain and process excess nitrogen
	Soil infiltration capacity	mm	Annual summed infiltration capacity of water
Air quality	Deposition velocity of Nox	cm/s	Capacity of ecosystems to capture and remove air pollutants
Recreation	Recreation potential index (RPI)	N/A	Capacity of ecosystems to provide recreational services

Table 5.1.: List of the Ecosystem Services and corresponding indicators used in the study (adapted from Maes *et al.* , 2011). HANPP data were obtained from Haberl *et al.* (2007).

assess the state of ecosystems and their services by 2014, addressed by the Working Group and Mapping and Assessment of Ecosystems and their Services (MAES). The results of this action will help inform policymakers but also contribute to the assessment of the economic value of ecosystem services which are to be integrated into the accounting and reporting systems at both EU and national level by 2020 (European Commission, 2011b).

The spatial mapping of ecosystem services throughout Europe forms the framework for the first part of this study. A total of 7 ecosystem services, represented by 9 indicators, were considered for this analysis (Tab. 5.1). In order for each ecosystem service to contribute equally to the analysis, and following the method of Petter *et al.* (2013), we standardized the data by reclassifying each service into a quantile split, producing a range of scores from 1 to 5 (five meaning high supply of a specific service). We then summed the 9 indicators to produce a map of “total” ecosystem services supply across Europe (Fig. 5.1.A). We used the HANPP (Human

Appropriation of Net Primary Production) data presented in Haberl *et al.* (2007), as the indicator for food provision. The HANPP values were only extracted within agricultural land as to not repeat the information on the provision of timber.

Our findings show a non-surprising spatial correlation between the supply of services and the European land-cover (Fig. 5.1.B). Low stocks for ecosystem service supply appear mainly around urbanized and densely populated areas and in arable land, e.g in central and eastern Spain, Southern Romania, Eastern UK, and Denmark. However, low total supply of services doesn't mean a low quality of the supply of individual services. For example, even if food production were at their highest level in some areas, if that is the only services provided, such area would appear in the low range of the map. High ecosystem supply occurs mainly in pastures, forests, and (semi) natural areas, such as the North-west Iberia, Scandinavia, central France, and central Romania.

Overall, we also observe that key areas of ecosystem service supply in Europe coincide with mountain regions (Fig. 5.1.A), mainly consisting of forest and (semi)-natural areas (Fig. 5.1.B). As a matter of fact, dense forest cover in mountain areas, and regions rich in wetlands (including mountain and lowlands) have been previously assessed as regions of high ecosystem service supply (Maes *et al.* , 2012a). Nonetheless, this does not mean that non-mountain areas do not supply valuable ecosystem services. For example, intensively managed agro-ecosystems are generally associated to flatter more fertile regions in Europe and considered essential food source providers.

Changes in human demand for services associated with specific land uses have shown diverging trends in Europe, varying between regions. In general the demand for crops, timber (mainly in northern countries), freshwater, and recreation has increased in the last 50 years while livestock production and wild foods supply have followed a decreasing trend throughout much of Europe's rural areas (Harrison *et al.* , 2010). During this period the quality of some ecosystem services have improved, mainly those services associated to forest ecosystems in mountain systems (i.e. timber production, freshwater provision, erosion and natural hazard regulation, and recreation), partly due to a decrease in human pressure in remote areas of the continent (Harrison *et al.* , 2010).

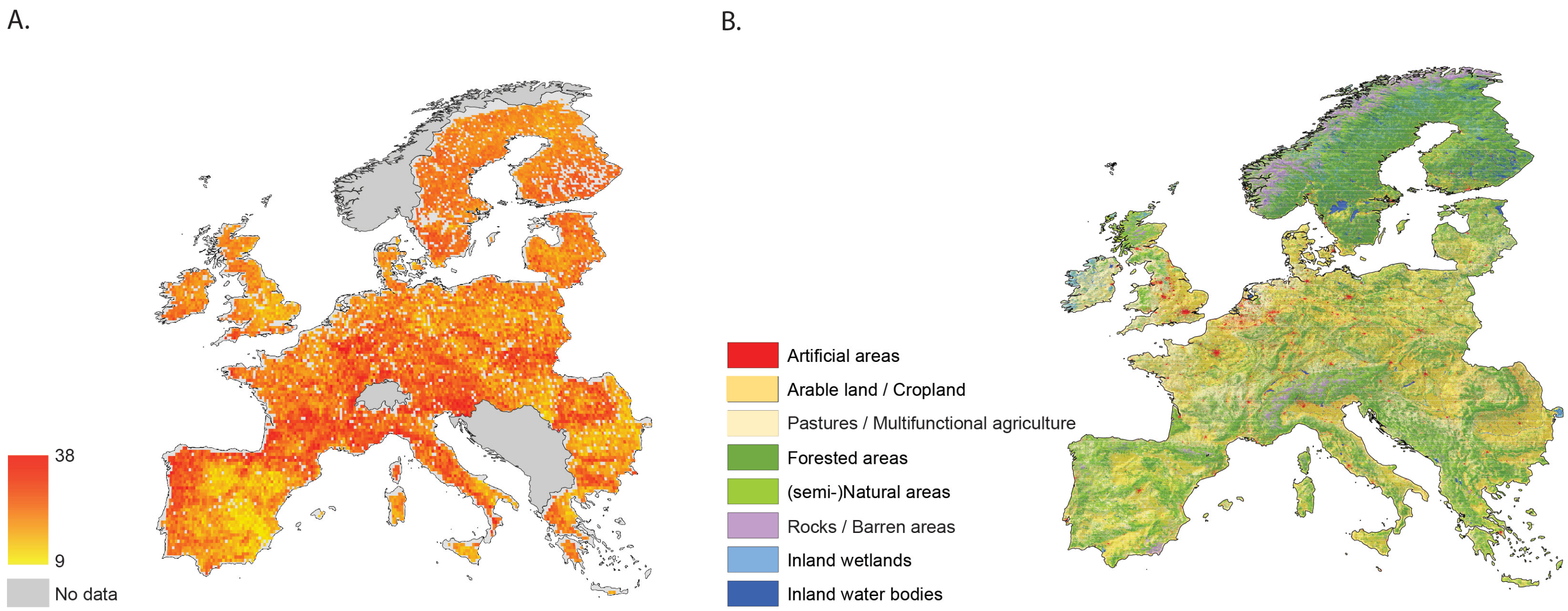

Figure 5.1.: Ecosystem services and land-covers in Europe. A. Sum of the quantile splits of all indicators used in the analysis. With each quantile split, services can reach values between 1 and 4, 4 being the highest. By summing all 9 indicators (see Tab. 5.1), the gradient potentially varies between 9 and 63 but, de facto, the maximum and minimum values are 9 and 38. (Method detailed in the text – source: Maes *et al.* , 2011); B. Map of European land-covers based on the Corine Land Cover data base of 2006 (EEA, 2010a). The map shows the elevation below the land-cover.

Threats and opportunities

Society has along the years taken for granted the services that are provided by natural ecosystems, exerting pressures due to human demand. This has lead to the unsustainable use and degradation of their natural functions and processes, disregarding the fact that ecosystems are fundamental in providing goods and services essential to human well-being and economy (TEEB, 2010). The Millennium Ecosystem Assessment (MA, 2005) was the first scientific appraisal of the global condition and trends of ecosystems and their services. The assessment revealed that in the last 50 years, increased anthropogenic pressures have changed ecosystems globally to meet rapidly growing demands for provisioning services, such as, food, freshwater, timber, fiber, and fuel. This led to a decline in other ecosystem services, mainly regulating services (e.g. flood regulation and nutrient cycling). According to the MA (2005), approximately 60% of the ecosystem services examined are being degraded or used unsustainably and expected to increase in the context of climate change. In Europe, the state of most ecosystem services has been categorized as degraded or mixed. However, from 1990 to present, timber production associated to forests and mountains, freshwater provision, natural hazard regulation and recreation have all improved (Harrison *et al.* , 2010).

Land conversion has been identified as the primary driver of ecosystem change (Klein Goldewijk *et al.* , 2011). Other drivers of substantial influence include climate change, nutrient application to agricultural systems, biological invasions and diseases, as well as indirect drivers such as socio-economic, political and demographic changes (MA, 2005). Years of unsustainable farming practices and mismanagement through agricultural intensification have resulted in a highly fragmented Europe, impacting biodiversity and the supply of ecosystem services (Schröter *et al.* , 2005; Egoh *et al.* , 2008). In particular, the unsustainable management of agro-ecosystems has contributed to the loss of habitat and biodiversity, soil erosion and nutrient runoff (Dunbar *et al.* , 2013).

In an attempt to halt further degradation or loss of ecosystem services and biodiversity, Europe has adopted the new Biodiversity Strategy Plan for 2020. The new 2020 strategy has a focus on ecosystem services, highlighting their natural and economic value and the importance of maintaining and restoring them (European Commission, 2011b). The EU Strategy delineates a total of six Aichi Targets which aim at reducing pressures on biodiversity and ecosystem services, and mapping and assessing the state of ecosystems and their services to incorporate their full value into national and EU accounting and reporting systems. For instance, target 2 (Action 6) promotes the restoration and the use of green infrastructures (i.e interconnected network of ecosystems, such as wetlands and woodlands) through incentives, restoring 15% of degraded ecosystems through both EU funding and Public Private Partnerships (European Commission, 2011b).

Acknowledging that both ecosystems and the services they provide are at risk due to human pressures, and that humans rely directly and indirectly on them for their well being, sets the frame for both a paradox and an opportunity, depending on which service is referred to. On the one hand, some services put ecosystems at a risk of overexploitation and unsustainable use, while, on the other hand, the supply of other services can become an incentive to preserve or restore ecosystems. Through the restoration of green infrastructures, natural areas can be reconnected, thus improving ecosystem functionality. The introduction of financial incentives is also seen as a cost-effective measure of investing in nature through the protection and restoration of services, reducing costs related to treatments and potential natural hazards (de Groot *et al.* , 2013). For example, the Danube river wetland restoration project, has promoted the concept of payment for ecosystem services (PES) to restore ecosystem capacity to retain flood waters, decreasing flood impacts, ultimately reducing flood costs while improving benefits to nature, people and local economies (IPCDR, 2010). Besides its growing incorporation in government policies, the use of PES (payment for ecosystem services) has also been applied at the business level. For example, the Nestlé Waters Programme, has created a union with local farmers in northeastern France to adopt farming practices that reduce nitrate pollution, providing high quality drinking water while compensating farmers (Bishop & Timberlake, 2007).

Though this type of economic incentives make sense in inhabited areas, where farmers, for example, could be asked to apply agri-environmental schemes to promote biodiversity and ecosystem services, a different approach can be envisioned in areas where the land is progressively relieved from agriculture and abandoned (Merckx & Pereira, in review). Over the last couple of decades, some European landscapes have witnessed the transition from extensive agriculture and semi-natural grasslands to abandoned land. This phenomenon is driven by several factors, which include the lack of economic opportunities, shifts in values and attitudes among people, a decreased dependency of local provisioning services (e.g. food and fuel) and environmental constrains, amongst others (MA, 2005; MacDonald *et al.* , 2000; Pereira *et al.* , 2005). Along the years these drivers have crossed paths, joining forces, exacerbating rural exodus, leading to land abandonment, for the most part in marginal regions (see chapter 2).

Although not a recent phenomenon, this rapid land use change has caught the attention of conservationists, economists, scientists, and policy makers. However, there still remains a large disagreement between those who see abandonment as a threat and those who see this trend as a window of opportunity for those ecosystem services which in the past were relinquished due to human demand. These contradictory viewpoints create challenges in delineating land management plans as well as adequate policies. In the past, policy measures, such as the Less Favored Areas (LFA) have been implemented in an attempt to attenuate rural migration and promote agricultural activity in less productive and remote regions of Europe (Dax, 2005). For instance, approximately 92% of the total mountain areas in the EU27 have been designated as LFA (EEA, 2010b).

However, efforts have gone unnoticed, which is clearly represented by the continuous decreasing trend of agricultural area and rural population (see chapter 2), both projected to continue in the future. Presently, only 14% of the total farmers in the EU 25 benefit from compensatory payments under the LFA (European Commission, 2008c), while, the new proposal for the Common Agricultural Policy (CAP) has included the restoration and conservation of ecosystem services under the rural development pillar (European Commission, 2011c). Considering that past attempts have been fruitless in halting abandonment in remote and less productive rural systems, and funded on the socio-economic challenges of maintaining extensive agriculture active, we can now look at the opportunity of rewilding these areas throughout Europe as a mean of generating environmental, social and economic benefits from ecosystem services provided by restored and self-sustainable ecosystems. By using present wilderness areas as a proxy we can determine the potential supply of ecosystem services through the promotion of rewilding in those areas projected to undergo abandonment.

5.3. Wilderness and Ecosystem services

Wilderness

Wilderness areas have been defined as large natural areas, unmodified or slightly modified governed by natural processes, with no human intervention, infrastructure or permanent habitation present (Wild Europe, 2012; Ceausu *et al.* , in press). In Europe, core wilderness areas are mainly concentrated in high altitude areas, predominantly mountain areas. Nordic mountains represent the highest proportion (28%) of wildest areas, followed by the Pyrenees (12%) the eastern Mediterranean islands and Alps (9 %), and British Isles (8 %) (Carver, 2010). However, remnants can also be found throughout much of the continent, where anthropogenic interference has slightly altered the natural ecological conditions (Carver, 2010). Note that, though the concept of wilderness is now commonly understood by the scientific community in Europe, the definition of wilderness will depend on the metrics chosen (Ceausu *et al.* , in press) and, as a result, its spatial distribution can vary from one study to another. Currently, there are several maps on potential wilderness in Europe, however, in the present study we chose to use Carver's (2010) quality wilderness index.

Many wild areas are under threat and represent a small proportion of the European continent. For example, although forests make up 33% of Europe's land cover, only 5% (9 million hectares) is considered wild. Currently, numerous organizations are focusing on the expansion of wild areas through the restoration of adjacent areas, while protecting remaining pristine regions. PAN Parks was one example of how the creation of a network has resulted in the protection of wilderness throughout Europe (http://www.panparks.org).

Wild ecosystems are healthy systems that provide a wide range of ecosystem services. They are stable and self-sustainable, able to maintain their structure, function and resilience over time (Costanza & Mageau, 1999). They play an important role in protecting services such as, air quality, freshwater provision, and supporting wildlife, including charismatic species, such as bisons and bears, that are reliant on wilderness areas (Russo, 2006; Helmer *et al.* , in press). Wild ecosystems also have the capacity to supply higher quality services than other types of systems. For example, there is higher carbon storage capacity in undisturbed forest, peatland and wetland (Schils *et al.* , 2008), subsequently providing additional environmental benefits (e.g.

biodiversity, water storage and water quality).

Moreover, wilderness areas provide a range of social and economic benefits. Several programs have integrated the use of wild areas to address urban issues such as youth at risk, youth development and rehabilitation (Hill, 2007), and recognized as a cost-effective form of healthcare. In addition, wilderness inspires educational programs (e.g. Jobse *et al.* , in press). Wilderness areas also provide spiritual benefits, such as, solitude, places of inspiration, a calm environment, and recreation/tourism (Heintzman, 2013; Ewert *et al.* , 2011). These social benefits can give birth to employment opportunities and thus generate income. For example, the Oulanka National Park in Finland brings €14 million per year to the local economy and employs 183 individuals (Huhtala *et al.* , 2010).

Mapping methods

We used Carver's (2010) wilderness quality map and the same quantile approach described earlier in the previous section to produce a gradient of wilderness quality with qualitative values ranging between 1 and 4 (4 meaning the highest wilderness quantile and high supply of ecosystem services). We then proceed to grouping the ecosystem services into provisioning, regulating, and cultural services and followed the same splitting approach for each category of service. The ecosystem services maps were then overlaid with the wilderness map. To determine the relationship between gradients of both ecosystem services supply and wilderness quality, we display the overlay of high and low wilderness with high and low supply of ecosystem services (Fig. 5.2.A.,B.,C. and D.). Furthermore, we used the projections of the CLUE model (Verburg & Overmars, 2009) to assess the potential change in the provision of ecosystem services with scenarios of land abandonment and rewilding in Europe for 2030. We considered as potential land abandonment and rewilding the cells classified as arable land, pasture, irrigated arable land, permanent crops in 2000 and classified as (semi)-natural vegetation, forest, recently abandoned arable land and recently abandoned pasture land in 2030 common to all four EURURALIS scenarios. For quantitative comparisons, we calculated the mean provision of ecosystem service (per km^2) in agricultural areas (based on the 2000 land use map, in Verburg & Overmars, 2009), in the top 5% high quality wilderness, and in the areas currently under agricultural use but projected to become abandoned by 2030, in the Iberian Peninsula. Significant differences between the distributions of the values for each type of land-use were tested using a Kruskal Wallis test. Finally, the ratio between the average supply of each indicator, and the common highest value for these indicators, were calculated in all three land-uses type studied in order to compare the relative supply of ecosystem services in each case. All mapping and data extraction were done using ArcGIS version 10.3, while the statistical analysis was done using R version 2.15.3.

Qualitative Analysis

We determined that some of the high wilderness areas (see Color Key 3 & 4 on Fig. 5.2.A) are associated to those regions, mainly mountain systems, supplying high ecosystem services (see CK 4). The same qualitative analysis was done separately for provisioning, regulating, and cultural services (represented by an indicator of recreational services).

In comparison with other categories of ecosystem services, the overlay of provisioning services and wilderness (Fig. 5.2.B) presents relatively large areas of high supply of services and low wilderness (see CK 2, e.g. in France, Benelux and Germany), along with areas of low service supply and high wilderness (see CK 3, e.g. Northern Scandinavia). This is not surprising since wilderness areas are typically associated with low to no extraction of natural resources. Southern Scandinavia provisions a number of important services, such as cereal production (Kettunen *et al.* , 2012), and is also more densely populated than in the north. There are nonetheless some representations of high provisioning services in areas of high wilderness quality (see CK 4), mainly associated to mountain regions (e.g. some areas of the Alps and Apennines). This can be due to the occurrence of large quantities of resources for some provisioning services (i.e. timber and freshwater) in mountain regions, which are also wilder than the rest of Europe.

The spatial distribution of regulating services coincided relatively well with wilderness (Fig. 5.2.C), with areas of Europe containing both high supply of services and high degrees of wilderness (see CK 4, e.g. Northern Iberia, Austria). Most of the continent is still represented by areas of both low regulating services and low wilderness (see CK 1, e.g. Eastern UK, Poland), which also coincides with agricultural areas (Fig. 5.1.B). Interestingly, several areas of high service supply and low wilderness appear on the map (see CK 2, e.g. Western France and Ireland). This is mainly due to high values for the Net Ecosystem Productivity, and soil infiltration (to a lesser extent), in those productive areas that are not classified as wilderness (particularly pastures, on Fig. 5.1.B).

Finally, for recreational services (Fig. 5.2.D), we found a predominance of either areas of low service and low wilderness (see CK 1), or areas of high wilderness and high service (see CK 4). The pattern for areas of low wilderness and high service supply (see CK 2) is completely different than for the other categories of services, with a rather small representation on the map. We also observe a consistent amount of areas with high wilderness but low recreation potential (see CK 3). This particular result may be due to the challenges associated to measuring the capacity and flow of benefits related to cultural services. For example, one may have an ecosystem of extreme beauty or wilderness quality, however, if they are not accessible, the flow of recreation and other cultural services is low. On the other hand, one may have a less natural area but easily accessible due to distance to human infrastructures such as roads. These, somehow contradictory, metrics can explain the observed pattern in the case of cultural services.

Taken as a whole, regulating and cultural service are often associated to high wilderness areas (Fig. 5.2.C and .D), particularly mountain systems. Mountain ecosystems cover approximately 41% of Europe's territory, providing various services due to their multifunctionality. They are generally referred to as "water towers" important for lower elevation ecosystems, irrigation, industry, hydropower and supply freshwater to more than half of the human population (Viviroli *et al.* , 2007). In mountain systems one can find a high proportion of habitat types with favorable conservation status (EEA, 2010b), playing a key role in provisioning many ecosystem services and maintaining ecological processes (Harrison *et al.* , 2010). These mountain habitat types include natural grasslands and mountain peatlands contributing to flood prevention, soil erosion, climate stability, and recreational services, such as bird watching (Silva *et al.* , 2008). Specifically, peatlands store large quantities of carbon(Hugron *et al.* , 2013) and play a fundamental role in climate regulation and are critical for water regulation. Meanwhile, grasslands are habitat to a large number of species, such as wild pollinators (Kremen *et al.* , 2002), which makes them essential in underpinning biodiversity and ecosystem services. These ecosystems' adaptive capacity to both temperature and altitude gradient have made them ecological hotspots of biodiversity and endemic species. In Europe, the highest number of endemic species can be found in the Alps and the Pyrenees (Väre *et al.* , 2003).

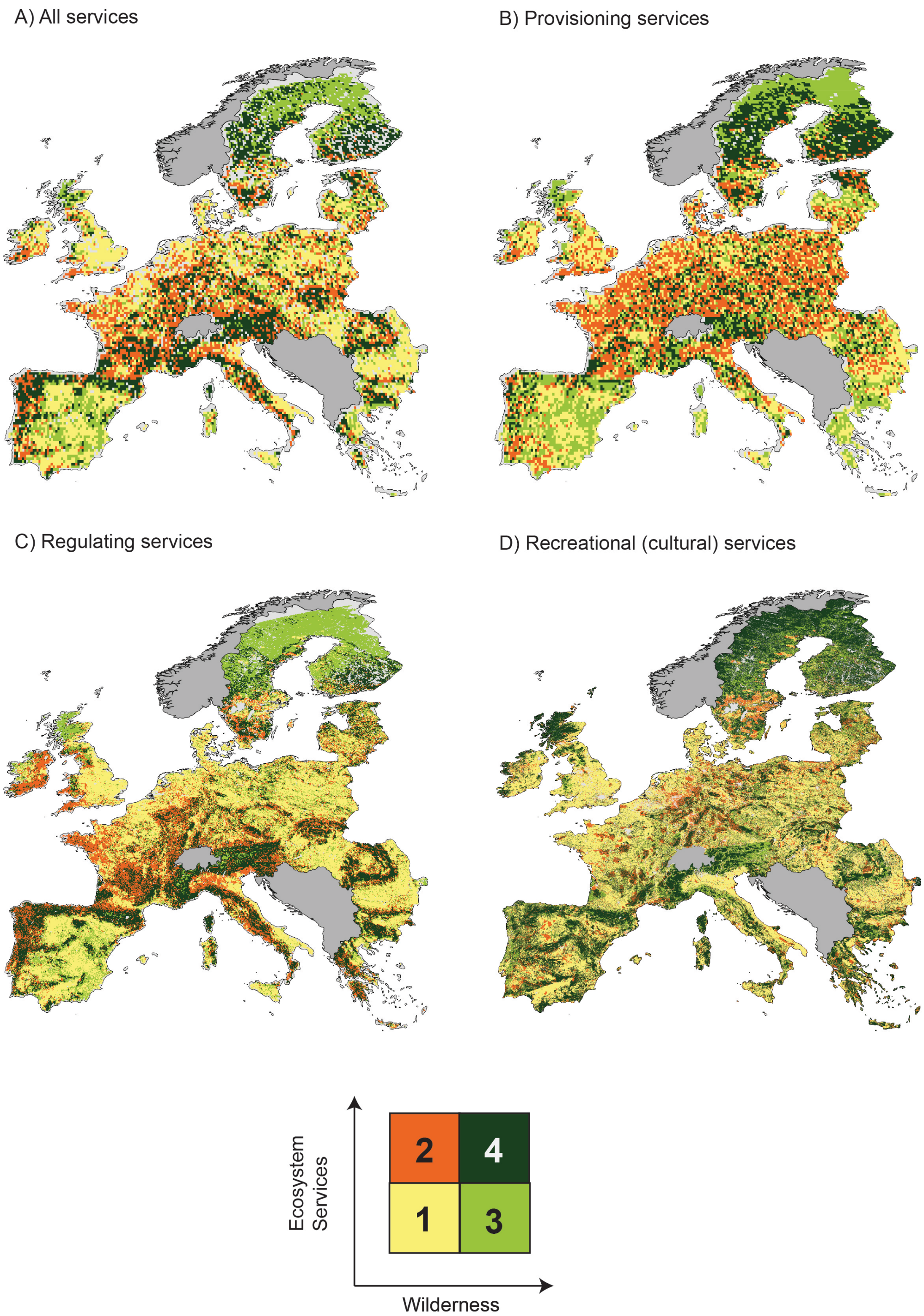

Figure 5.2.: Ecosystem services and wilderness in Europe (see method in the text). The values were grouped into "low" (bottom 50%) and "high" (top 50%) for both metrics and then joined, e.g. low supply of services and low wilderness (see color key on the figure). **A.** Indicators for all services; **B.** Indicators of provisioning services; **C**. Indicators of regulating services; and **D.** Recreational service. (See Tab. 5.1 for a details of the indicators used). Sources: Maes *et al.* (2011); Carver (2010).

Mountain systems also play a key role in regulating erosion and natural hazards influenced by vegetation cover (Körner *et al.* , 2005), more specifically forested land. Forests make up 41% of mountain systems (Körner *et al.* , 2005) and can be regulators of natural disasters as the soils of mature forests have high infiltration rate, thus reducing peak flows and floods (Maes *et al.* , 2009). They also provide a range of services such as carbon sequestration, air quality regulation, timber for fuelwood and non-timber products (game and medicinal plants), and climate regulation (Harrison *et al.* , 2010; Maes *et al.* , 2012a). Moreover, forests and mountain systems supply cultural services, holding spiritual and religious value to local inhabitants, and are main recreation and ecotourism attraction (Price *et al.* , 1997). Though mountain systems are tagged as major suppliers of ecosystem services, they also are fragile systems which recover slowly or not at all from disturbances which consequently decreases ecosystem function, impacting lowland areas and overall, human well-being (Körner *et al.* , 2005).

5.4. Rewilding and ecosystem services

Ecosystem services and scenarios of rewilding

Although, many reports have outlined the positive benefits of wilderness and wild areas, the potential gains and losses of ecosystem services through the promotion of rewilding is still understudied. Our analysis contributes to reducing this lack of data by quantifying the biophysical potential of a system to produce benefits after rewilding has occurred, by comparing it with the current supply in other agricultural areas and using wilderness as a proxy for the future supply of services. The analysis was performed at the extent of the Iberian Peninsula. Generally, we speculate that by increasing the size and connectivity of high quality wilderness, we will have an increase in the supply of ecosystem services associated with those habitats (that is, if the abandoned land is restored to a self-sustainable state, either naturally or via assisted restoration, see Rey Benayas & Bullock, in press and chapter 4). Concomitantly, the supply of services associated with agricultural land-uses might decrease.

By comparing the distribution of the values, at the extent of the Iberian Peninsula, for the supply of indicators of ecosystem services, between “areas currently cultivated", "areas cultivated but predicted to be abandoned", and "areas within the top 5% of Iberian wilderness", we were able to predict a potential increase or decrease in the supply of the studied services.

We determined significant differences in the supply of ecosystem services and the different land use intensities (Tab. 5.2). We found HANNP values to be significantly higher in present agriculture areas than in both land projected to be abandoned and top 5% wilderness areas. HANNP has been identified as an indicator measuring changes in biomass flows in ecosystems and the provision of important ecosystem services, as a result of land use change (Erb *et al.* , 2009). Our results are not unexpected as human production of food is greater in agricultural areas than all other land uses.

For deposition velocity of NOx, an indicator of air quality, values were higher in wilderness in comparison to the other land use intensities, land projected to become abandoned and present agriculture. Typically, trees perform better air quality regulation (Maes *et al.* , 2011). We can thus assume that air quality improves in wilderness areas due to the higher capacity of wild forested ecosystems to capture and remove air pollutants. Increasing wilderness areas should thus lead to increases in air quality.

Nitrogen retention was highest in wilderness followed by agriculture and land projected to be abandoned, while soil infiltration capacity was greatest in future abandoned land, followed by wilderness. Both nitrogen retention and soil infiltration are indicators of water regulation. We can assume that water regulation improves with increased vegetation dynamics, as high values for these indicators were mainly represented in both top wilderness and projected abandoned land.

Both timber and recreation values where highest for wilderness and areas projected to be abandoned. Nonetheless, these results may be misleading, specifically for timber, since values calculated for wilderness only include the top 5% of wild regions which is expected to be highly protected. Another explanation for the high values of timber supply in the top 5% wilderness may be due to the lower resolution of the data.

For carbon stock and net ecosystem productivity we found a higher supply in land projected to be abandoned than those areas designated as wilderness (Tab. 5.2). Thus, the abandonment of those areas would support wilderness by increasing its occupancy of productive areas, measured both in carbon stock and net ecosystem

Category	Service	Indicator	Unit	Agricultural Area	Top 5% high wilderness	Projected to be abandoned	p value
				Mean per km² (sem)			
Provisionning	Food production	HANPP	gC/m2/yr	279.03 (0.21)	224.12 (2.65)	271.15 (2.91)	<.0001 (***)
	Timber	total stock	m^3/ha	0.81E+05 (0.21E+03)	2.10 E+05 (1.67E+03)	1.34E+05 (1.68E+03)	0.23 (NS)
	Freshwater	Surface water flow	mm	151.05 (0.15)	156.08 (0.58)	244.77 (1.61)	<.0001 (***)
Regulation and Maintenance	Climate regulation	Carbon stock	ton/ha	24.82 (0.06)	63.19 (0.30)	74.81 (0.49)	<.0001 (***)
		Net Ecosystem Productivity (NEP)	mg/m^2/yr	5.13E+05 (0.34E+03)	6.99 E+05 (1.75E+03)	7.36E+05 (1.92E+03)	<.0001 (***)
	Water regulation	Nitrogen retention	%	2.69 (.00)	3.04 (.01)	2.46 (.02)	<.0001 (***)
		Soil infiltration capacity	mm	9.61 (0.02)	15.99 (0.11)	32.15 (0.23)	0.0006 (***)
	Air quality	Deposition velocity of Nox	cm/s	0.07 (0.00)	0.42 (0.00)	0.28 (0.01)	<.0001 (***)
Cultural	Recreation	Recreational Potential Index (RPI)	N/A	0.22 (2E-04)	0.43 (11E-04)	0.26 (17E-04)	<.0001 (***)

Table 5.2.: Quantitative analysis of the supply of ecosystem services in the Iberian Peninsula, on agricultural land, top 5% high wilderness areas and land currently cultivated and projected to be abandoned by 2030 (see Mapping methods section).

productivity, thus playing a fundamental role in climate regulation. Similar findings were found for freshwater provision, with higher values in land projected to become abandoned than on wilderness areas, values fairly close to present agricultural areas.

These results support the premise that releasing land from agricultural activity is an opportunity to increase or maintain high stocks of ecosystem services (Fig. 5.3). Not surprisingly, agricultural land in Iberia is mainly dedicated to food production (with HANPP as an indicator). Yet, the land currently cultivated but projected to be abandoned is an important supplier of carbons stocks and has high net ecosystem productivity and soil infiltration capacity (Fig. 5.3). Future abandoned lands also display high contribution in the provision of freshwater, which is another argument towards their passive restoration, rather than maintaining human activities on them. These results show that areas currently cultivated but projected to be abandoned have the potential to contribute to human well-being, both in terms of the quantity and the quality of the ecosystem services they can supply, which is probably due to their biophysical properties (e.g. altitude, quality of the soils). Nonetheless, the top 5% wilderness areas in Iberia also supply a wide range of ecosystem services (Fig. 5.3), witch reinforces the idea that increasing the area of wilderness via rewilding will promote the provision of high quality services.

Although these results exemplify positive trends for the majority of the indicators, they are to be read with caution as the transition from "recently abandoned" to "rewilded" is not fast, simple, or even guaranteed (see chapter 2 and chapter 4). Thus the supply of ecosystem services during the early stages after land abandonment can be hard to predict and complex to understand. A recent study revealed that decreases in land use intensity, primarily the abandonment of mountain grasslands, lead to initial decreases in net ecosystem exchange of CO_2 (Schmitt *et al.* , 2010). The impacts of nitrogen storage following abandonment are also not well understood. Nitrogen storage in younger grasslands is known to be lower than older grasslands (Deng *et al.* , 2013). Nonetheless, we can speculate that the benefits of releasing land from agriculture outweigh those of present agriculture activity. Recent studies have confirmed that the complexity of an ecosystem, which includes but is not limited to its vegetation dynamics, the age and the distance and the extent of fragmentation, are all elements which influence the supply of ecosystem services and biodiversity (Vanacker *et al.* , 2014; Ferraz *et al.* , 2014; Grêt-Regamey *et al.* , 2014). By restoring habitats, we would also increase the landscapes' multifunctionality, through its services.

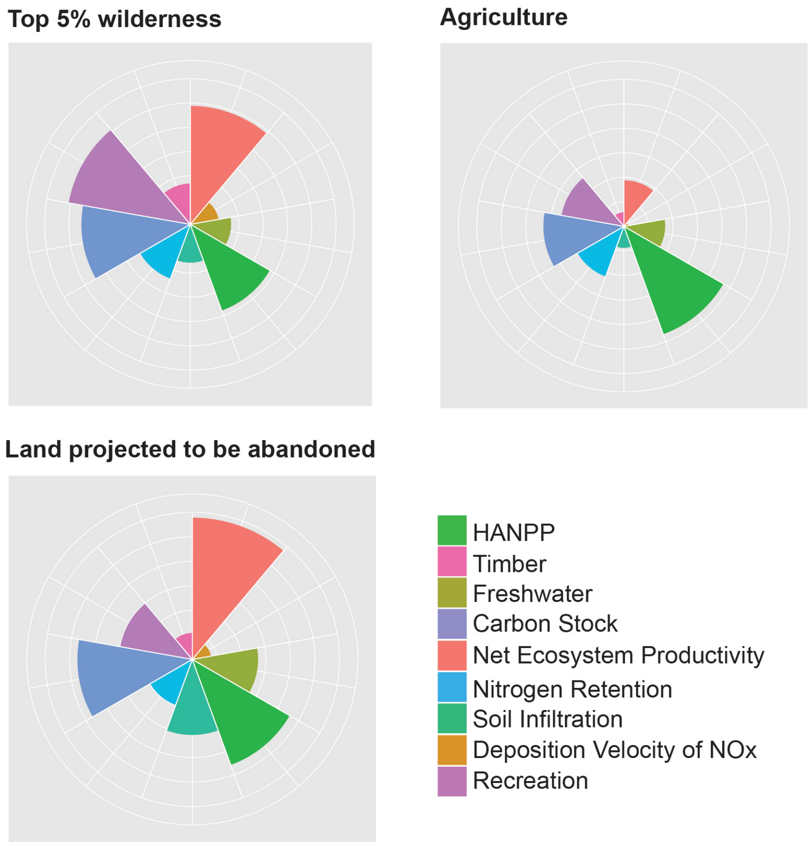

Figure 5.3.: Relative supply of ecosystem service per land category. For each land type (top 5% wilderness, agricultural area, and land projected to be abandoned), the diagram represents the average supply per km^2 relative to the maximum value of the entire sample (all land categories together). See Tab. 5.2 for the average values.

This analysis is based on projections of supply of service on areas where land abandonment or rewilding could potentially occur within the decades to come and so far is geographically limited to the Iberian Peninsula. Yet, rewilding has already occurred in some areas of Europe, either naturally or with assisted restoration and it is worth looking into the supply of ecosystem services in those "pilot" areas, though information on their benefits are still scarce. The earliest documented case of rewilding pioneered in the Oostvaardersplassen wetlands in the Netherlands, in the 70s (Vera, 2009). The introduction of wild herbivores, such as the red deer, Heck cattle and Konik ponies naturally substituted the ecological function of the now extinct aurochs and tarpans (Birdlife International, 2011; Vera, 2009). Today, it is one of Europe's leading wetland regions, where ecological restoration increased natural areas creating a win-win situation promoting regulating, cultural services and biodiversity while generating economic benefits through the promotion of tourism. However, there are still some limitations to this project as the perseverance of its success has been attributed to on-going human intervention along the years. In this section we present an additional number of case studies in which the promotion of wild areas and/or rewilding has generated environmental and economic benefits for local communities, landholders and society in general.

Regulating Benefits

The implementation of rewilding with forest regeneration on abandoned farmland can also be a major contribution to the mitigation of climate change through the sequestration and storage of carbon. In Lowland England, studies on different land use management options have shown that the cost and benefits of changes in ecosystem services from rewilding outweigh those from arable and dairy farming (NERC, 2012). In the Upland UK estimates show that managing the land for carbon storage and sequestration through the restoration of peatlands may be more profitable than pastoral activities (Reed *et al.* , 2013). As a matter of fact, peatlands, in Scotland, have been valued between €49 million to €196 per annum for carbon sequestration (McMorran *et al.* , 2006).

Likewise, it has been estimated that within the Natura 2000 network, commercial and wild forest habitats generate the highest carbon value estimated at €318.3 and €610.1 billion, in 2010, followed by grassland systems ranging between € 105.6 and €196.5 billion (ten Brink *et al.* , 2011). In the Carpathians, the protection of old growth forest will generate a funding of €26 million through carbon offsets, providing regional economic relief (ten Brink *et al.* , 2011). In the Hoge Veluwe Forest, a protected area of the Netherlands, total economic benefit generated by forests is €2000 ha/year, for the following services; wood production, supply of game, groundwater recharge, carbon sequestration, air filtration, recreation and nature conservation. This value is calculated to be three times higher than adjacent agricultural land (Hein, 2011).

The benefits of improving water quality upstream impacts lowland areas and human well-being (Forslund, 2009). Although there is still a lack of available information on the economic value of water purification at the EU level, studies confirm that cities such as Berlin, Vienna, Oslo and Munich benefit from the natural treatment from ecosystems in protected and non protected areas, with annual economic benefits ranging between €7 and €16 million for water purification and €12 and €91 million for water provision per city (ten Brink *et al.* , 2011). In the archipelago of the Azores, the restoration of pastures to native forests would result in an economic benefit of €110 thousand per year from water purification (Cruz & Benedicto, 2009). These examples, though limited, demonstrate that protecting and restoring natural vegetation is of economic interest, and should be integrated under the Water Framework Directive.

Floodplains (wetlands) are also important ecosystems, acting as natural sponges that retain water in river basins, slowly releasing down river and into groundwater. Moreover, they play a fundamental role in filtering out pollutants and are home to protected wildlife. A recent projected scenario revealed that restoring the function of floodplains in EU countries would be saving approximately €1.4 billion of treatment costs for water purification and reduced annual cost of flood damage, currently at €6.4 billion and expected to increase (Feyen & Watkiss, 2011). Of course, this type of restoration has initial costs. The Danube Basin restoration project estimates that the recovery of 100,000 ha, would cost 500,000 €/km^2, i.e. an investment of €500 million. However, this value is still estimated to be much lower than the costs associated to damage control and the improvement of dykes (WWF, 2010). Restoration of natural river landscapes has also been estimated to contribute to flood mitigations. In Belgium the Kalkense Meersen has calculated these benefits between €640,000 to €1,654 per annum (Arcadis, 2011).

Degradation of natural ecosystems has also been linked to the intensification of natural hazards (Dudley *et al.* , 2010). For example, in the Swiss Alps the protection of old forests contribute to disaster prevention (e.g. avalanches and landslides) and have been analyzed at a value of € 1.6–2.8 billion per year (IPCDR, 2010). Restoration of natural river landscapes has also been estimated to contribute to flood mitigations. Soil erosion control is another ecosystem service that can potentially benefit from rewilding, playing an important economic role in the prevention and/or mitigation of land degradation and desertification. The role of pristine scrublands against soil erosion was valued at € 44.5/ha, in 2008 throughout Europe and in Belgian grasslands (Ruijgrok & de Groot, 2006).

Cultural Benefits

Economic benefits from non-extractive activities such as nature tourism and recreation boost local and regional economies, providing income and employment to communities and private landholders who face limited alternative livelihoods, especially in a context of rural depopulation of marginal areas (Brown *et al.* , 2011; McMorran *et al.* , 2006). Most importantly, the aims of eco-tourism are closely associated with biodiversity conservation. Through the promotion of rewilding efforts, we can speculate that there will be a decrease in fragmented landscapes, creating a break for large mammals and other species (Russo, 2006), and indirectly increasing tourism while generating economic benefits to local communities.

Presently, eco-tourism is the fastest growing component sector in tourism (Gössling, 2000). Eco-tourism has constantly increased by 20-34% per year since the early 1990's (Bishop *et al.* , 2008). According to the International Ecotourism Society (TIES), eco-tourism involves the responsible visiting to natural areas that conserves the environment and improves the well-being of local people. Wildlife areas appeal to a large spectrum of tourists given the presence of charismatic species and other rare or attractive species. For example the reintroduction of wolves in the Yellowstone National Park has attracted additional tourists, generating economic and social benefits estimated at US$6-9 million per year (Donlan *et al.* , 2006). The reintroduction of ungulates and large carnivores in the Majella and the Retezat National Park in Italy and Romania, respectively, has successfully contributed to the local economy (Kun & van der Donk, 2006). In Scotland, tourism from wild landscapes is one of the most important economic sectors, contributing €1.6 billion, annually, to the country´s economy. In particular, recreation opportunities, such as wildlife watching and hillwalking, generate €65 million and support 39,000 full time jobs (Bryden *et al.* , 2010; Brown *et al.* , 2011).

The reintroduction of the beaver, though controversial at the time of its implementation, is thought to potentially generate an additional €2.4 million per year into the local Scottish economy through eco-tourism (Campbell *et al.* , 2007). In addition to its potential economic benefits, beaver dams are considered to have a positive impact on river systems by increasing both invertebrate and fish populations (Kemp *et al.* , 2010). Although many remain uncertain to beavers positive impacts on nature, quantitative evidence has revealed that the reintroduction of the species results in an increased habitat diversity and abundance of fish, specifically salmon (Kemp *et al.* , 2010; BSWG, 2013).

Safeguarding the Romanian Carpathians Ecological Network is a success case study on rewilding initiatives improving the quality of life of those who live there through the development of local economies while focusing on the conservation of natural values and cultural heritage (Maanen *et al.* , 2006). The Carpathian Mountains and the Danube Delta are considered the biodiversity and wilderness hotspots of Romania. In Zarnesti, a small community in Romania increased their total local revenue from € 140,000 in 2001 to € 260,000 in 2002 through eco-tourism programmes (CLCP, 2000) The Natura 2000 network further exemplifies another cost effective mean of protecting wildlife while generating benefits. Annually, the gross socio-economic and co-benefits (social and environmental) from the Natura 2000 network range between €223 billion and €314 billion, representing between 2 and 3% of EU´s GDP (ten Brink *et al.* , 2011). This figure contrasts with the annual investment in the Natura 2000 network, estimated at €5.8 billion (Gantolier *et al.* , 2010) while providing 8 million (FTE) jobs (ten Brink *et al.* , 2011). In other words, rewilding could locally develop a new economy around the use of wilderness through the creation of new markets, such as eco-tourism (Helmer *et al.* , in press).

5.5. Discussion

The degradation, or land conversion, of natural ecosystems alters not only species richness and composition; it reduces ecosystem functionality, impacting the flow of ecosystem services, the costs of recuperation and ultimately human well-being (Flynn *et al.* , 2009). Agriculture in Europe has now taken two different paths, the marginalization of agriculture in rural mountain landscapes and the intensification in regions with more fertile soils (MacDonald *et al.* , 2000; Strijker, 2005). The abandonment of extensive agriculture, mainly in mountain areas, has been a result of various drivers leading to rural exodus. Years of combating rural desertification and the maintenance of agricultural practices through incentives has not contributed to the attenuation of this phenomenon (Merckx & Pereira, in review).

The management of these abandoned lands has become a challenge for many conservationists, as policies and government strategies have recently integrated the restoration of ecosystem services alongside the conservation of biodiversity. According to the CBD Global Biodiversity Outlook for 2010 the opportunity for restoring nature across Europe through rewilding is now (SCBD, 2010). The restoration of nature through rewilding is seen as a solution to the on-going land abandonment, developing a bold new economy while offering social and environmental benefits, based on the restoration and sustainability of ecosystem services provided by wildlife, wild areas and wilderness (McMorran *et al.* , 2006; Hein, 2011; Gantolier *et al.* , 2010; Donlan *et al.* , 2006; Bryden *et al.* , 2010; Brown *et al.* , 2011). A recent study has also performed a cost-benefit analysis of restoration projects and determined that ecological restoration results in positive investments (de Groot *et al.* , 2013).

In this analysis, we investigated the existence of a spatial co-occurrence of a gradient of both wilderness and ecosystem services supply (Fig. 5.2). In addition, we looked into the impacts that rewilding efforts post land abandonment could potentially have on the supply of ecosystem services. Overall, we found positive indicators that high degrees of wilderness co-occur with a high supply of ecosystem services (Fig. 5.2.A). This spatial co-occurrence appears even stronger when looking into regulating and cultural services (Fig. 5.2.C and .D). Furthermore, our results provide quantitative evidence that the opportunity of restoring abandoned land to a self-sustained natural state (rewilding) could increase the supply of regulating and cultural services in Iberia (Tab. 5.2).

We thus argue that by restoring and sustaining wilderness areas we are underpinning a supply of high quality ecosystem services provided by wild areas. In other words rewilding can be used as a conservation management tool and restore the provision of ecosystem services. For example, the expansion of wilderness areas will play an important role in mitigating climate change through the sequestration of carbon. These services will also heighten a new economy market, based on services supplied by wilderness, providing an economic break for viable rural communities through the creation of jobs and income generated from incentives (e.g. PES, carbon markets and eco-tourism). However, the implementation of incentives in remote areas faces limitations when rural depopulation and land abandonment have already occurred.

Although, the concept of rewilding is fairly recent in Europe, it has already been identified as a cost-effective management strategy for traditional land uses in Scotland (McMorran *et al.* , 2006; Brown *et al.* , 2011). In the Netherlands, the promotion of rewilding has been positively perceived: individuals attribute a low willingness to pay for the conservation of extensive farming versus rewilding initiatives, which were generally ranked high in terms of attractiveness (van Berkel & Verburg, 2014). However, we cannot generalize rewilding as the only cost-effective strategy, positively perceived by all rural inhabitants.

From a holistic perspective, active, assisted, and passive restoration promoting wilderness needs to be viewed as a management strategy that provides benefits to humans through the flow of ecosystem services, while preserving biodiversity. Nonetheless, the application of these two management strategies is dependent on the state of the ecosystem (Chazdon, 2008), the climatic, biophysical, socio-economic elements of the location, as well as the costs and benefits associated to each management option. Thus, the total eradication of human intervention needs to be pondered. Semi-natural grasslands rely on adequate management regimes, through grazing activity. In Europe, decreases in pasturing activity, has lead to the natural encroachment of vegetation, reducing landscape heterogeneity, impacting those species associated to open spaces and resulted in the loss of functional groups (Laiolo *et al.* , 2004; Lindborg & Eriksson, 2004; Peco *et al.* , 2012). Consequently, protecting these ecosystems involves the reinstatement of natural disturbances regimes, safeguarding ecosystem processes (see chapter 4). Another limitation with rewilding is the time to restoration, depending on the type and the degree of intervention that is implemented. Natural forest regeneration in Europe, for example can take from 20 years to nearly a century (Verburg & Overmars, 2009), depending on the cultivation history and on the geographical conditions. In these cases, the return on investments can be long, before the supply of ecosystem services starts increasing.

We are not suggesting that rewilding efforts through assisted or passive restoration be the only solution to Europe's present situation, but be considered as a potential strategy in those areas where the social-ecological dynamics of the landscape are no longer socially, economically and environmentally sustainable. Yet, there are still many challenges in understanding the full relationship between landscape management, the supply of ecosystem services, the economic benefits and costs associated to each management type, and finally how they can be framed into wilderness areas and adopted in environmental policies (Jobse *et al.* , in press).

There are still many questions to be answered concerning rewilding efforts throughout Europe. However, we argue that the promotion of the restoration of those regions undergoing abandonment is an opportunity for ecosystems services and biodiversity. By protecting and increasing wilderness we are underpinning areas of high ecosystem service supply. Notwithstanding, several pitfalls and trade-offs can be associated to rewilding. We have yet to determine how the promotion of rewilding would affect social-ecological systems, primarily humans' ability to adapt to changes in the provision of ecosystem services. Another consequence to rewilding is the potential loss of traditional cultural values and heritage while its social and environmental implications are still unknown (Cerqueira *et al.* , 2009).

The emerging balance calls for further research and increase awareness of the environmental, social and economic benefits associated to wilderness areas. Raising awareness of these benefits may help to promote the concept and reinforce the idea of how naturalness is an opportunity for increasing overall human well-being and defining public policies and funding of nature conservation policies.

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6. Towards a European policy for Rewilding

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“A society is defined not only by what it creates, but by what it refuses to destroy.”

John Sawhill

Laetitia M. Navarro conducted the research and the redaction of the manuscript. Henrique M. Pereira contributed to developing and discussing the ideas presented in the chapter.

Resumo:

Milhões de hectares na Europa que anteriormente foram utilizados para a agricultura estão na iminência de se libertarem da permanente influência humana nas próximas décadas devido a um crescente abandono da actividade agrícola. O conceito de *rewilding*, o retorno ao estado selvagem, para estes terrenos que progressivamente são deixados ao abandono, representa uma oportunidade de restauração, não só da paisagem que integram mas também da biodiversidade e serviços de ecossistema que destes provêem e que até então estavam restringidos ao parco numero de áreas selvagens existentes no velho continente. Dada esta realidade, a Europa necessita de rapidamente definir e adoptar uma estratégia política que promova o *rewilding* e a sua implementação como uma medida alternativa de gestão para a Conservação. Necessita ainda também de uma metodologia para avaliar os resultados e de um mecanismo de partilha e promoção de boas práticas entre os diferentes agentes envolvidos.

Neste capítulo, apresentamos primeiramente a história das políticas de conservação e áreas protegidas na Europa. Discutimos, mais detalhadamente, o que se entende por áreas designadas Áreas Protegidas Nacionais (NPD) e como a sua percepção tem vindo a mudar ao longo do tempo. Estando inicialmente associado maioritariamente a um nível de protecção integral, o estatuto das NPDs evoluiu progressivamente ao longo do tempo no sentido de permitir a presença e intervenção do ser humano e suas actividades dentro dos limites das NPDs. A comprovar esta tendência note-se o facto de que, apesar da maioria do estados membros possuírem actualmente mais de 18% do seu território classificado como área protegida, na sua maioria poucos são os casos cujas áreas protegias (mais de 3% dessas áreas) pertencem à classificação da IUCN de tipo I e II, direccionadas para conservação da natureza. O sistema de classificação de áreas protegidas da IUCN é um sistema internacional que classifica as áreas protegidas em 6 tipos diferentes. Os diferentes tipos estão classificados segundo o objectivo principal da área protegida em questão, o que por conseguinte influencia o plano de gestão da mesma e o nível de intervenção humana permitida.

Áreas protegidas de tipo I e II têm como mote principal a conservação da Natureza e por essa razão possuem o mais elevado nível de restrições à presença humana.

Neste capítulo apresentaremos também a adopção da Directiva Habitats e Pássaros, assim como o processo de implementação da rede Natura 2000. Este último é um exemplo único de uma rede de áreas protegidas regional e além-fronteiras. Um aspecto interessante é o facto de as práticas de gestão dos vários sítios Rede Natura 2000 usarem as paisagens europeias tradicionais como ponto de referência para a conservação, deste modo atribuindo relevância ao papel desempenhado pelas terras agrícolas de baixo impacto na conservação destas áreas. Em 2013, mais de 23 000 sítios na Europa continental haviam sido atribuídos a classificação de Natura 2000. Em muitos países, os sítios designados Natura 2000 coincidiam em simultâneo com áreas protegidas nacionais.

O papel desempenhado pela Conservação nas actividades dos diferentes sectores, nomeadamente na agricultura, é também abordado neste capítulo. Ainda relativamente ao papel da Conservação na agricultura, este é discutido através do conceito das Áreas Agrícolas de Elevado Valor para a Natureza (*High Nature Value Farmland*) e da implementação dos programas agro-ambientais subsidiados pela Política Agrícola Comum (CAP) da União Europeia. A eficiência das medidas referidas é analisada e discutida. Analisamos com especial atenção a pretensão de que a CAP pode potenciar positivamente a biodiversidade, tendo em consideração que o primeiro e segundo pilar da CAP: preferência por práticas agrícolas intensivas e atribuição de subsídios que promovam práticas agrícolas amigas do ambiente, respectivamente, são contraditórios; e o facto de que a grande maioria de abandono agrícola ainda acontece em áreas remotas e de baixa produtividade.

Analisamos a crescente importância do papel atribuído às áreas selvagens (*wilderness*) e a necessidade de integrar políticas de gestão adequadas para as mesmas no contexto Europeu. Recomendamos que o aumento de áreas selvagens na Europa se faça através de processos naturais de *rewilding* de paisagens abandonadas, potenciando assim o estabelecimento de novas áreas selvagens na paisagem Europeia dotadas de estatuto de protecção legal e gestão adequadas. Deste modo, o processo de *rewilding* poderia contribuir para uma rede de áreas protegidas mais coerente e interligada nos 28 Estados Membros.

Neste capítulo avaliamos ainda a contribuição da criação de zonas selvagens e do *rewilding* como meio para atingir objectivos globais e Europeus. Por fim, propomos

algumas recomendações no sentido da inclusão do *rewilding* no contexto das políticas de Conservação Europeias.

Palavras-chave: áreas protegidas nacionais, Natura 2000, áreas agrícolas de elevado valor para a Natureza, programas agro-ambientais, áreas selvagens (*wilderness*), objectivos de Conservação, políticas de gestão territorial.

Towards a European policy for Rewilding

Abstract: Millions of hectares of agricultural land could be released from human pressure within the next decades in Europe. Rewilding presents a great opportunity to restore the abandoned landscapes, along with the biodiversity and the supply of those ecosystem services that were until now restricted to the remaining few wild areas of the continent. As a result, rewilding is in a dire need of a policy framework in Europe, to promote its implementation as a land management option, to evaluate its outcomes, and to share knowledge and good practices among stakeholders. In this chapter, we present the history of conservation policies and protected areas in Europe, the implementation of the Natura 2000 Network being one of the major milestones. The role of conservation in sectoral activities such as agriculture is also discussed. The growing importance given to wilderness areas and the inclusion of wilderness management into European policies is then presented. The chapter then evaluates the contribution of wilderness and rewilding to the achievement of global and EU targets. Finally, recommendations are made to efficiently and adequately include rewilding into the European framework of conservation policies.

Keywords*:* Nationally Designated Protected Areas, Natura 2000, High Nature Value Farmland, Agri-environmental schemes, wilderness, Conservation targets, land management policies.

6.1. Introduction: a historical perspective

Though evidence of land conservation goes back several thousands of years in Europe, the concept of protected areas was first implemented across the continent by the 15th century, when poaching and logging were banned from royal hunting forests by the nobility in order to protect the game (Jones-Walters & Čivić, 2013; Possingham *et al.* , 2006; Ramão *et al.* , 2012). Those protected areas (PAs) were designed to preserve a given resource (e.g. timber or game), rather than to preserve nature in general. It was not until the 19th century that landscapes would be preserved for their "natural beauty", following a movement initiated in Germany to preserve Naturedenkmal, i.e. nature monuments (Jones-Walters & Čivić, 2013). The first "National Parks" (NP) were designated in the USA, in Yosemite NP, in 1864, then Yellowstone NP, in 1872 (Possingham *et al.* , 2006), with the aim of preserving nature for recreational, cultural and ethical reasons (Borrini-Feyerabend *et al.* , 2013). The first European park was created in Sweden, in 1909 (Pinto & Partidário, 2012; Ramão *et al.* , 2012). In 1969 the IUCN gave an official definition to "National Parks" as the first resolution of its 10th assembly (IUCN, 1969).

The 1970s mark a change in the way PAs were managed in Europe, shifting from strict protection to the acknowledgment of the role and needs of local communities and other stakeholders, and their integration in the management of the landscape (Jones-Walters & Čivić, 2013; Ramão *et al.* , 2012). In 1971, the UNESCO launched the Man and Biosphere (MAB) program, leading to the concept of Biospheres Reserves in 1974 (Coetzer *et al.* , 2014). It was followed, twenty years later, by the establishment of the World Network of Biosphere Reserves (UNESCO, 1996), with a particular focus on the involvement of local communities, and their sustainable use of the resources present within the area. Today, the MAB network counts 167 reserves in the EU28, with more than a fourth (n=45) in Spain. 1971 is also the year of the signature of the Ramsar Convention, for the global cooperation and conservation of wetland habitats (Possingham *et al.* , 2006). The member states of the EU28 count 843 of those sites, for a total area of 13,4 million ha (ramsar.wetlands.org). In 1972, the UNESCO also signed the "Convention concerning the Protection of the World Cultural and Natural Heritage" (World Heritage Centre, 2013). The first EU Natural Heritage Sites were established in 1979, in Croatia (Plitvice Lakes National Park) and in Poland (Białowieża Forest). In 2013, the EU28 counted 27 "natural" and 6 "mixed" Heritage sites (whc.unesco.org).

In this chapter, we first present the status and trends of current biodiversity conservation in the European Union, via the national designation of Protected Areas, the Natura 2000 network, and agri-environmental schemes. The recent integration of wilderness in the EU conservation framework is then discussed, along with the potential of rewilding abandoned farmland. Rewilding and wilderness conservation are then evaluated in regards to the achievement of the global and European conservation targets set for 2020. This chapter only discussed continental conservation, and marine protected areas were removed from the analysis.

6.2. Current conservation policies in the EU

Nationally Designated Protected areas

Nationally Designated Protected Areas (NDPAs) encompasses a variety of designations: “national park”, “regional park”, “nature park”, “nature reserve”, “biosphere reserve”, “wilderness area”, “wildlife management area”, “landscape protected area”, and “community conserved area” (Dudley, 2008; Ramão *et al.* , 2012), which also vary greatly in their management policies. When countries are divided into federal states (e.g. Spain, Germany), each entity can also have regional designation policies. Moreover, some countries protect specific ecosystem nation-wide (e.g. wetlands in Croatia, rivers in Portugal), without designating them within their protected areas (Ramão *et al.* , 2012). More than 31% of the European NDPAs cover forest ecosystems, while agro-ecosystems are represented in over 28% of the areas (Ramão *et al.* , 2012). These areas also tend to be designated in mountain regions, due to their remoteness and the resulting lower human densities. The NDPAs are classified by the IUCN into six protection categories (Dudley, 2008), defined in 1994, based on the level of management and the allowed degree of human activity (sec. 6.2), though not all areas are yet classified, or even registered as such. Out of the 68% of NDPAs classified by IUCN categories in Europe (N= 52 995), the vast majority belongs to category IV, Habitats/Species management areas (sec. 6.2). However, category V (Protected landscape/seascape) covers the largest area on the continent. It is also interesting to notice that the most strictly PAs (Categories I and II) are not the most representative, both in terms of number and area, with coverage of 20% of the total protected areas. Nonetheless, although comparatively few areas are in category II (National Parks), they cover an area almost similar to the most represented type of protected area, category IV (respectively 88155 km^2 and 88352 km^2 in sec. 6.2).

The historical distribution of the different types of NDPAs matches the history of the European perception of protected areas. From the 1950s to the mid-1960s about half of the PAs were in the most restrictive categories (mostly national parks, Cat.II),

Category	Name	Management type	Detail	Number (%)	Total Area in km^2 (%)
Ia	**Strict nature reserve**[1]	Strict Protection	Strictly protected areas set aside to protect biodiversity and also possibly geological/geomorphological features, where human visitation, use and impacts are strictly controlled and limited to ensure protection of the conservation values. Such protected areas can serve as indispensable reference areas for scientific research and monitoring.	4514 (6%)	14549.18 (2%)
Ib	**Wilderness area**	Strict Protection	Protected areas are usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, which are protected and managed so as to preserve their natural condition.	1207 (2%)	34672.43 (5%)
II	**National park**[2]	Ecosystem conservation and protection	Protected areas are large natural or near natural areas set aside to protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area, which also provide a foundation for environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.	320 (<1%)	88155.57 (13%)
III	**Natural monument or feature**	Conservation of natural features	Protected areas are set aside to protect a specific natural monument, which can be a landform, sea mount, submarine cavern, geological feature such as a cave or even a living feature such as an ancient grove. They are generally quite small protected areas and often have high visitor value.	3124 (4%)	4571.65 (1%)
IV	**Habitat/species management area**	Conservation through active management	Protected areas aim to protect particular species or habitats and management reflects this priority. Many category IV protected areas will need regular, active interventions to address the requirements of particular species or to maintain habitats, but this is not a requirement of the category.	**31654 (41%)**	88352.17 (13%)
V	**Protected landscape/seascape**	Landscape/seascape conservation and recreation	A protected area where the interaction of people and nature over time has produced an area of distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values.	10837 (14%)	**319117.34 (47%)**
VI	**Protected area with sustainable use of natural resources**	Sustainable use of natural resources	protected areas conserve ecosystems and habitats, together with associated cultural values and traditional natural resource management systems. They are generally large, with most of the area in a natural condition, where a proportion is under sustainable natural resource management and where low-level non-industrial use of natural resources compatible with nature conser- vation is seen as one of the main aims of the area.	1339 (2%)	35044.49 (5%)
	Not classified	N/A	N/A	24420 (32%)	97781.40 (14%)

(1) Two protected areas were asigned to category I, without distinction and were not counted in this table.
(2) Areas designated as "National parks" in Europe can fall in different IUCN categories than II.

Table 6.1.: Description of the different IUCN Categories for protected areas and contribution of the Nationally Designated Protected Areas of Europe to those categories (Dudley, 2008; EEA, 2013a).

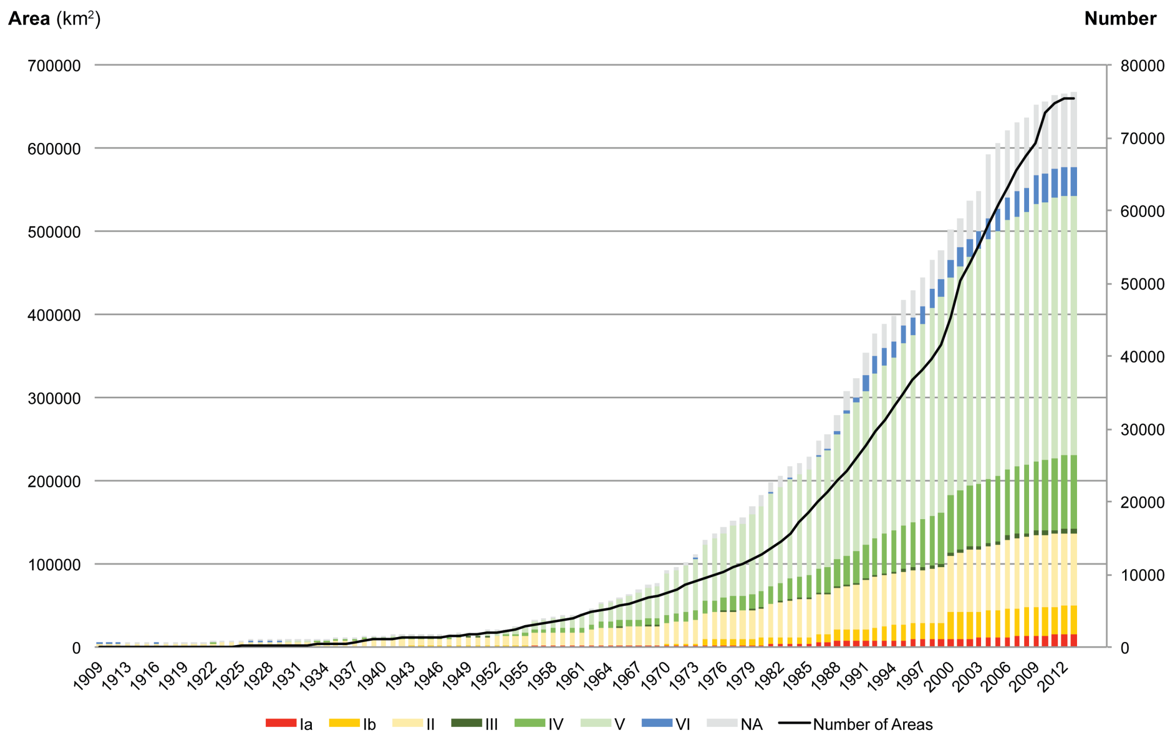

Figure 6.1.: Temporal evolution of the number of Nationally Designated Protected Areas in Europe, and the total area protected. The NDPAs are classified according to the IUCN categories, NA meaning that the area was not yet classified (EEA, 2013a).

while the other half were managed with the inclusion and/or tolerance of human activity (Fig. 6.1). In the 1970s there was a large increase of PAs designated as the less restrictive category V (Protected landscape). Currently, the IUCN categories II and I represent less than a quarter of the total classified PAs of Europe (Fig. 6.1).

Birds and Habitats Directives

The Council of Europe signed the Bern Convention in 1979 to give a legal framework for the conservation of biodiversity on the continent (Jones-Walters & Čivić, 2013). This was followed by the adoption of the Bird Directive by the 9 member states of the EU in April 1979 (79/409/EC), to respond to worrying decreases in bird populations observed on the continent, and acknowledging that some species of birds are a European heritage and that the conservation of migratory species is a trans-boundary matter. The Directive was later amended, as new member states joined the EU and was updated in November 2009 for the EU27 countries (Directive 2009/147/EC).

The directive's articles engage the member states, inter alia, into maintaining populations at viable levels, creating protected areas, and up keeping and managing bird populations within and outside those areas. Particular attention should be given to bird species in Annex I (193 species), while species in Annex II (82 species) may be hunted under national legislations. The directive resulted in the creation of Special Protection Areas (SPAs), which number increased steadily, including with the addition of new member states to the European Union.

The Bird Directive was followed, 13 years later, by the Habitat directive (92/43/EEC), adopted in May 1992. This directive emphasizes the conservation of biodiversity via the conservation of "habitats, wild fauna and flora", in a context of sustainable development for the continent. A total of over 230 habitat types and over 1000 species of animals and plants were selected, in order for countries to suggest lists of Sites of Community Importance (SCIs), which were then evaluated by the Commission, before being implemented as Special Areas of Conservation (SACs) by the member states (European Commission, 2002; Gaston *et al.* , 2008b). The Habitat Directive further aimed at building a "coherent European ecological network", the Natura 2000 Network (N2000), which would encompass the PAs created under the Bird Directive of 1979, the SPAs, and the newly designated SACs. The contribution of each EU country to the N2000 network was set to depend on the proportion of habitats (in annex I) and habitats for species (in annex II and IV) present within their borders.

The management of the N2000 areas is the responsibility of each member states, which can delegate and decentralize to federal or regional agencies (European Commission, 2002). Traditional European landscapes may serve as a conservation baseline (Gaston *et al.* , 2008b), as the guidelines on N2000 site management emphasis the importance of ensuring "the continuation of traditional management regimes, which very often have been critical in creating and maintaining the habitats which are valued today" (European Commission, 2002).

The N2000 network is a unique example of a regional, transboundary, and unified network of protected areas (Crofts, 2014; Hochkirch *et al.* , 2013). As of 2008, Denmark and the Netherland had reached 100% of their sufficiency for the Habitat Directive Annex I habitats and Annex II species, meaning that their network of PAs covered at least one instance for 100% of the habitats and species of the annexes that had a known distribution on their territories (EEA, 2009b). The rest of the EU member states had between 70% and 99% of sufficiency, with the exception of Lithuania (61%), Czech Republic (59%), Cyprus (25%) and Poland (17%). The N2000 Network is now shifting from establishing the areas to defining proper coordinated management strategies (European Commission, 2013).

Overall picture of protected areas in the EU

The ensemble of protected areas in the European Union, composed of Nationally Designated Protected Areas (NDPAs), Special Areas of Conservation (SACs), and Special Protection Areas (SPAs) is extensively covering the continent (Fig. 6.2.A). As of 2013, the EU28 counted over 77 000 terrestrial NDPAs and nearly 23 000 continental Natura 2000 areas. Yet, 30% of the area protected in Europe represents an overlap between a type of designation or another. As a matter of fact, in some countries, such as Spain, Slovenia, and Estonia, the N2000 areas almost entirely overlap with NDPAs (Fig. 6.2.A). At the European scale, the overlap is particularly true for NDPAs in the IUCN categories I to IV (Ramão *et al.* , 2012).

The majority of the Member States count more than 18% of their territories in a protected area (Fig. 6.2.B). Nonetheless, the map of Europe depicts a different picture when focusing on the most restrictive conservation categories of the IUCN (Categories I and II on Fig. 6.2.C): most countries have less than 3% of their area in those categories. Sweden, Belgium, and Slovakia are the only countries protecting more than 5% of their national area as a strict nature reserve, a wilderness area, or a National Park (Fig. 6.2.C). N2000 areas overlapping with NDPAs classified as categories Ia and Ib represent 4% of the network (European Commission, 2013).

The EU Protected areas tend to be created in high and remote areas, with lower productivity (Dudley, 2008; Gaston *et al.* , 2008a), and with less regard for the habitats and the species that inhabit them than for the availability of the land. Nonetheless, conflicts might arise with local populations when an area used for resource extraction is set to be strictly protected. Such tensions are exacerbated by strictly top-down approaches, i.e. with the lack of consultation of local stakeholders in the establishment of a PA, which is often the case with the establishment of N2000 areas (Crofts, 2014). On the contrary, less restrictive categories, or “multiple use” PAs are typically more easily accepted (Possingham *et al.* , 2006).

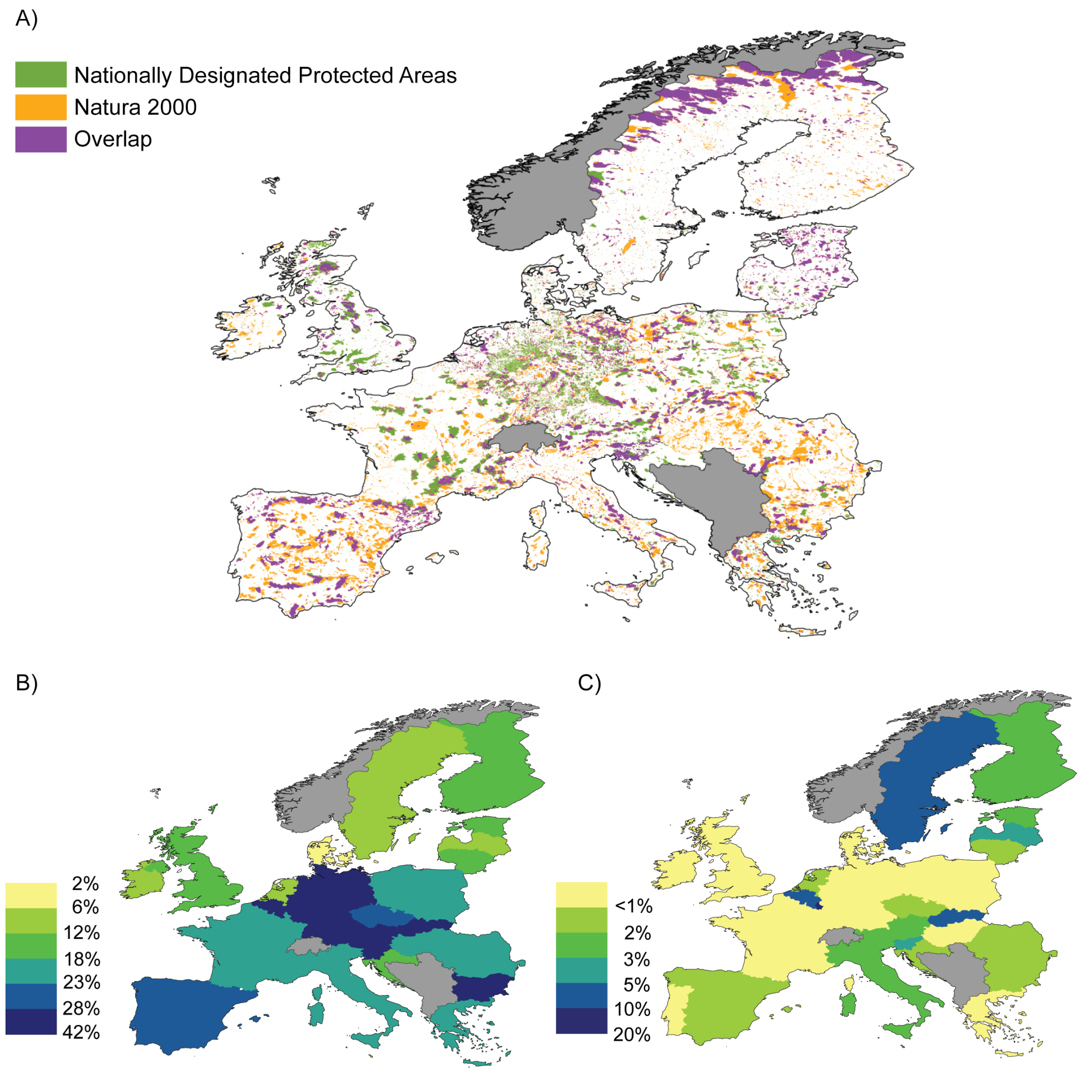

Figure 6.2.: Spatial perspective on European protected areas (EEA, 2013b,a). **A)** European network of Nationally Designated Protected Areas, Natura 2000 sites, and overlap between the two designations; **B)** Proportion of each EU28 country within a protected area (Nationally Designated Protected Areas and Natura 2000 sites); **C)** Proportion of each EU28 country within a protected area in category I or II of the IUCN.

Moreover, designating a protected area is one thing, but establishing it in situ and managing it efficiently will depend on the financial and political supports of the local governments (Leverington *et al.* , 2010; Pinto & Partidário, 2012). As a result, designated PAs might suffer from a lack of adequate monitoring budget and trained staff (Hochkirch *et al.* , 2013). The N2000 Network has also recently been criticized for its lack of flexibility, adaptability, and monitoring (Crofts, 2014; Hochkirch *et al.* , 2013).

6.3. Agriculture and conservation

Extensive agriculture is often associated with rich biodiversity in European conservation (EEA, 2004; Halada *et al.* , 2011). As a result, “High Nature Values Farmland” (HNVF) areas were designated in the 1990s and now cover 15 to 25% of the EU countryside (EEA, 2004). In particular, some of the N2000 sites are covered on more than a fourth of their area by extensive farmland (EEA, 2004). HNVF is characterized by its dependence to human activities to create and maintain it, by blocking the process of natural successions (EEA, 2004; Halada *et al.* , 2011; Merckx & Macdonald, in press; Merckx, in press). In a review of the 231 habitats types of the Annex I of the Habitat Directive, Halada et al., (2011) identified 63 habitats depending on agricultural practices for their management, 23 of which are “fully dependent”, while 40 “partly depend” on agriculture, mainly due to the prevention of natural successions.

HNVF areas are currently threatened by two opposing forces, intensification of agriculture on one side, and rural depopulation and farmland abandonment on the other (EEA, 2004, 2009a). In 2003, following the Kyiv Resolution on Biodiversity, HNVF in Europe were officially to be identified and preserved (EEA, 2009a), using the Common Agricultural Policy (CAP) as a tool for their conservation. It was integrated into the second pillar of the CAP, along with Less Favored Areas (LFAs). Agri-environment schemes (AES) and other EU subsidies thus became the mechanism of HNVF conservation (EEA, 2004).

Additionally, though the European Parliament recently stated that the EU biodiversity policies were not well integrated into sectoral policies as energy, transport, and agriculture (European Parliament, 2009), agri-environmental policies have attempted for quite some times to better integrate agricultural productivity and biodiversity conservation. Currently, EU funds address the relationship between farmers and conservation in two ways. On the one hand, the EU compensates farmers receiving a lower income due to environmental restrictions. On the other hand, the EU created incentives for farmers to develop an environmentally friendly agriculture. Both forms of subsidies are not exclusive. Following the 2003 amendment of the regulation on Rural Development of the EU (1783/2003), farmers will receive monetary compensations for the "costs incurred and income foregone" resulting from the classification of their land as a N2000 site according to Article 16 (1). Articles 22 to 24 of the same regulation directly address AES, and how "support should be granted to farmers who give agri-environmental [...] commitment for at least five years" (Article 23). The subsidies are destined to cover the "income foregone", "additional costs resulting from the commitment", and "the need to provide an incentive" (Article 24). The payment of subsidies and the implementation of agri-environmental policies vary greatly from one Member State to the other (EEA, 2009a).

Nonetheless, the consequences of the subsidizing nature conservation through the CAP are debatable. A first contradiction can emerge when the first pillar of the CAP favors intensive and productive agriculture on one hand, and hence fragments natural habitats (Crofts, 2014; EEA, 2009a; Henle *et al.* , 2008), while on the other hand the second pillar incents farmers to develop environmentally friendly practices. Additionally, the compensations paid to farmers in LFAs (supported by the second pillar of the CAP to limit farmland abandonment) poses no real limits to intensification and overgrazing, provided that farmers follow country-specific "good farming practices" (EEA, 2004). Finally, there is no direct link between the amount spend in CAP subsidies per ha and the level of HNVF in an area (EEA, 2009a, 2004; Halada *et al.* , 2011), or between the occurrence of HNVF and the designation of Natura 2000 sites (Henle *et al.* , 2008). The second contradiction results from the inadequacy of CAP subsidies in remote and less productive areas. The phenomenon of rural depopulation was initiated in the 1950s in Western Europe, driven by socio-economic factors interacting to create a "circle of decline" in those remote areas (MacDonald *et al.* , 2000; Rey Benayas *et al.* , 2007), which is not likely to be interrupted, despite of the rural development policies that have been implemented, and the resulting payment of subsidies (see Fig. 2.3 in chapter 2).

The direct consequence of the phenomenon of rural depopulation is the abandonment of farmland in the less productive areas of the EU (see chapter 2). Agricultural land abandonment is typically perceived negatively in developed countries (Queiroz *et al.*

, 2014; Meijaard & Sheil, 2011), as a result of, inter alia, observed land encroachment, increased risk of fires, and decreases in populations of farmland birds. Yet, the withdrawal of human activities from those areas is also an (often disregarded) opportunity to increase the area of wilderness in the EU by applying rewilding as a land management policy.

6.4. Opportunities for wilderness and rewilding

Wilderness is both an ecological and social concept. The ecological meaning and extent of wilderness will vary depending on the metrics used (Ceausu *et al.* , in press), though the core of its definition is the (quasi) absence of human impact, the large size of the area (e.g. 10 000 ha), and the naturalness of the dynamics that govern the ecosystems (European Commission, 2013; Fisher *et al.* , 2010). The social and subjective concept of wilderness and wildlands is, for example, associated with the notions of remoteness and solitude (Fisher *et al.* , 2010; Fritz *et al.* , 2000). As a result, the definition of wilderness by the various people experiencing it will also depend on their perceptions of such areas (Nash, 1967). Probably one of the most famous definition was given by the Wilderness Act of 1964 in the United States as " [...] an area where the earth and its community of life are untrammeled by man, where man himself is a visitor who does not remain." (US Congress, 1964). By definition, characterizing an area as "wilderness" will condition its management to "no-intervention" or "set-aside" practices (European Commission, 2013). Despite a lack of consensus on its concept, scientists agree that Europe is one of the continents with the least amount of wilderness (Mittermeier *et al.* , 2003), mainly due to its long history of human induced land-use changes (see chapter 4). Currently, the wilderness of the EU28 is mainly located in Scandinavia and in mountainous areas (Ceausu *et al.* , in press and see chapter 5).

Globally, wilderness and protected areas do not necessarily coincide. Using human density, size of the area, and historical intactness as metrics, Mittermeier *et al.* (2003) found that only 7% of the world's remaining wilderness was included in PAs of IUCN categories I to IV. When looking into all types of PAs in Europe, there is little to no correlation between the location of NDPAs and Natura 2000 areas with higher values in the WQI (Fisher *et al.* , 2010). However, there is a correlation between the occurrence of areas under the IUCN Categories I and II and high wilderness quality (Fisher *et al.* , 2010). The number of NDPAs in Europe that falls under the IUCN Ib category ("wilderness areas") is rather low and spatially restricted: 1207 areas (i.e. 2% of the classified PAs) are concerned (sec. 6.2), and are located mainly in Sweden, Estonia, Slovakia, and Slovenia. Only 12 of the 28 EU member states manage PAs designed in categories Ia or Ib, with different national legislation regarding the designation, the size of the area, the type of management and the level of human activity allowed (European Commission, 2013).

Nevertheless, European wilderness is progressively gaining more importance, both in science, conservation policy and at their interface. Its role in halting biodiversity loss was officially recognized (European Parliament, 2009; Jones-Walters & Čivić, 2010), with a will to include wilderness in the post-2010 targets. As a result, the European Parliament called for an effort to define both wilderness and the benefits derived from it, and for a better integration of wilderness in conservation policies. A special care was to be given to wilderness areas within the N2000 network. Indeed, some conceptual conflicts can arise when the non-interventional management of wilderness areas goes against the management of secondary (semi-natural) habitats of Annex I, such as the "European dry heaths" and "Dehesas with evergreen Quercus spp." (Halada *et al.* , 2011), unlike primary habitats, which rely on natural processes, for example "Western Taiga" and "Bog woodlands" (European Commission, 2013; Fisher *et al.* , 2010).

The European Commission (2013) recently published guidelines on the management of wilderness areas within the Natura 2000 Network. Though not legally binding for the member states, they illustrate the will to include wilderness in EU conservation policies. The guidelines state that management practices for wilderness in the N2000 network can involve the total or partial interdiction of human activities. When applicable, zonation can be used to define an area of non-intervention management for the wilderness core habitat, and a managed zone for secondary habitats. The guidelines also emphasize the importance of addressing local communities, to explain them the functioning of non-intervention management, and the benefits they could derive from it. Finally, scale has its importance in the designation and management of wilderness areas, as too little, or too fragmented land would not meet the criteria to allow for natural processes (European Commission, 2013).

With the ongoing trends of farmland abandonment occurring on the continent, and the momentum gained by wilderness, rewilding appears as a viable land management option (see chapter 2). It consists in the restoration of ecological processes and self-sustaining ecosystems, either passively, or with low to mild levels of intervention if the land-use history requires it (Rey Benayas & Bullock, in press and see chapter 4).

Increasing the area of wild land via the rewilding of abandoned landscapes will contribute to delineating new wilderness areas in the European landscape, with adequate conservation status and appropriate management. As such, rewilding can further increase the ecological coherence and connectivity of the protected areas in the EU28. Increasing the area of wilderness via rewilding will also contribute to the large scale natural processes that maintain it (e.g. European Commission, 2013).

Some of the most emblematic species benefiting from land abandonment and rewilding are large mammals (Enserink & Vogel (2006); Russo (2006); Ceausu *et al.* (in press); Deinet *et al.* (2013) and see chapter 2). They demand a large availability of land in order to sustain their home range requirements (Jones-Walters & Čivić, 2010), and limit conflicts with humans, which also makes wilderness areas essential to their conservation (Mittermeier *et al.* , 2003; Ceausu *et al.* , in press). Additionally, species listed in the Birds Directive, which are specialists of old-growth forests (e.g. the three-toed woodpecker - *Picoides tridactylus*, the black stork - *Ciconia nigra*), or which have large habitat requirements (e.g. the Siberian tit - *Parus cinctus*, the black woodpecker - *Dryocopus martius*), can benefit from the increase of wilderness areas (European Commission, 2013).

The notion of a “perceived wilderness” (Jones-Walters & Čivić, 2010) is important when investigating the benefits supplied by rewilded areas for people. For example, the increase in wild areas and the resulting wildlife comeback are thought to contribute to reconnecting Europeans with nature (Deinet *et al.* , 2013). The cultural services provided via the enjoyment and experiencing of wilderness, for example, the perception of solitude and remoteness, can reciprocally motivate its conservation and guide policies and land management. Wilderness areas also supply a wide range of provisioning and regulating services, such as freshwater provision, carbon sequestration, and nitrogen regulation (see chapter 5). Having in mind the benefits of rewilding and increased areas of wilderness, we can now investigate which could be their contribution to Global and European targets.

6.5. Global and European conservation targets

After failing at meeting the biodiversity targets which had been set for 2010 (Butchart *et al.* , 2010), all parties of the Convention on Biological Diversity (CBD) adopted an agreement in Nagoya, which set 20 Aichi Targets to preserve biodiversity and ecosystem services by 2020 (CBD, 2011). Several targets can be addressed by protected areas, wilderness, and rewilding. In particular, target 11 requires that "at least 17% of terrestrial and inland water [...] are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas [...]". For most European countries, this target has already been reached, in the sense that most countries have more than 17% of their national territory within a protected area (Fig. 6.2.B), though the level of protection, and its effectiveness varies tremendously between areas and between nations (e.g. Fig. 6.2.C). For other tasks, the level of completion is not so easily measured. Target 15 calls for the enhancement of ecosystems' resilience including through the "restoration of at least 15% of degraded ecosystems" in each nation, and the increase of carbon stocks. Rewilding being a particular case of restoration, it can thus directly be linked with the achievement of this target, particularly when looking into the increases in carbon stocks that could result from an enlargement of wild areas (see chapter 5). Finally, target 12 requires the prevention of the extinction of threatened species and the improvement of their conservation status. Again, the rewilding of abandoned landscapes, and an increase in wilderness areas, can directly contribute to this target, as several species already show increasing trends (LCIE, 2004; Deinet *et al.* , 2013; Boitani & Linnell, in press and see chapter 2). Unrelated with PAs and wilderness but not so with rewilding, target 7 requires that "areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity", while target 3 calls for the termination, or the reform, of "incentives, including subsidies, harmful to biodiversity". Both these tasks could be addressed by a reform of the subsidies system of the CAP and the AES (e.g. Merckx & Pereira, in review).

The Aichi targets and their implications are not legally binding for countries. Nonetheless, the EU and all its Member States adopted them in the European Biodiversity Strategy and defined new regional strategies to 2020 (Tab. 6.2), in order to both halt biodiversity loss and restore degraded systems (European Commission, 2011a; Hochkirch *et al.* , 2013). Some of these targets can be addressed by an efficient, and when needed better designed, network of PAs. The preservation of wilderness

European targets	Status in 2010	Objective for 2020
1. Implement the habitat and bird directives	17% of habitats and species protected by the Habitat directive are in favorable status	34% of the habitats and 26% of the species should either improve or be in a favorable status.
	52% of the bird species are in a secure position	80% of bird species should be secured or improving
2. Maintain and restore ecosystems and their services	No continental data on degraded ecosystems, and the supply of ecosystem services	Increase knowledge and define actions - Mapping and assessment of the state of ecosystems and their services - Definition of a strategic restoration framework, including with the development of green infrastructures - Ensure no let loss of biodiversity and ecosystem services
3. Increase the contribution of agriculture and forestry to biodiversity	Only 15-25% of extensive high nature value farmland remains. 7% of the habitats and 3% of the species protected by habitat directive and depending on agriculture have a favorable status. Farmland bird populations have decreased by 50% since 1980 but have now leveled of Farmland butterfly populations have decreased by 70% since 1990.	Maximise agricultural areas covered by biodiversity measures of the CAP. - Enhance direct payments for environmental public goods in the EU CAP. - Better target Rural development to biodiversity conservation. - Conserve Europe's agricultural genetic diversity.
	21% of forest habitats and 15% of forest species protected under the habitat directive have a favorable status. 1-3% of forests are in natural and unmanaged status.	Forest Management Plans, in line with Sustainable Forests Management are in place for all publicly owned forest and forest holdings above a certain size. - Encourage forest holders to protect and enhance forest biodiversity. - Integrate biodiversity measures in forest management plans.

Table 6.2.: EU targets and biodiversity strategies to 2020, most relevant within the context of protected areas, wilderness and rewilding discussed in this chapter (European Commission, 2011a).

is also considered as playing a crucial role in reaching some of the targets (European Commission, 2013), namely "protecting and restoring biodiversity and ecosystem services" (targets 1 and 2), and "reducing pressures on biodiversity" (targets 3 and 5). Additionally, wilderness areas, being remote and not densely populated, present the advantage of lower land prices per hectare, while non- intervention implies drastically lower management costs (Mittermeier *et al.* , 2003).

The EU also incorporated the Aichi Target 3 to its plan, in order to "reform, phase out and eliminate harmful subsidies at both EU and member states level" (Target 6 - Action 17c), when such subsidies have been identified as having indirect negative consequences for biodiversity. Moreover, the Commission highlights the importance of integrating biodiversity policies into wider European policies concerns (e.g. agriculture, forestry), while maintaining them as lucrative activities (European Commission, 2011a). Furthermore, there exists a will to "minimize the duplication of effort and maximize synergies between efforts undertaken at different levels" (European Commission, 2011a). In a context of farmland abandonment in remote and less productive areas, maximizing the synergies between conservation efforts can be done by redirecting subsidies towards rewilding (Merckx & Pereira, in review; Merckx, in press), while allowing the remaining local population to live off the land through different means than its cultivations. An efficient implementation of rewilding for the management of the abandoned land will have, in the long run, a positive impact on biodiversity and the supply of ecosystem services (see chapter 2 and chapter 5). The latter includes cultural services, such as ecotourism, which will directly benefit local populations.

6.6. Recommendations for rewilding

The current European policy response to pressures on biodiversity can be either with site protection (e.g. SPAs, SACs), or with the regulation of the activities of those exploiting the land, which can also be relying on voluntary actions, i.e. with Agri- Environmental Schemes (EEA, 2004). Rewilding abandoned farmland can efficiently contribute to reaching European and Global conservation targets. But in order to do so, a policy framework must be designed to include rewilding in the land management options given to practitioners (see chapter 2). To that extent, European conservation policies must aim toward several goals.

In places where people still keep a strong link with nature, and particularly with wilderness, its come back via natural regeneration should not be problematic (McGrory Klyza, 2001). Yet, when the link with traditional landscape is the stronger, as in many regions of Europe, rewilding might be perceived negatively (Bauer *et al.* , 2009; Höchtl *et al.* , 2005). Communication between scientists, policy-makers, decision-makers, and the public will be essential to allow the implementation of rewilding, and to promote the values of wilderness in a landscape. Development initiatives are also known to ease the transitions between one form of management and another, by increasing the support of local communities for the protected area (Pinto & Partidário, 2012). Giving the opportunities to populations to shift their activities from low-income agriculture to ecotourism in rewilded areas can be an efficient way to meet both ends (Helmer *et al.* , in press and see chapter 5).

The proposed "greening" reform of the CAP could enforce the implementation of a system of compensations for stakeholders maintaining low productive practices in order to preserve traditional agricultural habitats (Hochkirch *et al.* , 2013). Another option is to maintain payments for farmers that apply environmentally friendly practices on productive soils, and redirect subsidies on less productive lands towards rewilding(Merckx & Pereira, in review). By doing so, member states will still be able to meet the demands for agricultural goods, yet promoting responsible and green practices on productive soils, while the lands left abandoned due to their remoteness, their lower productivity, and the difficulty to cultivate them (MacDonald *et al.* , 2000; Rey Benayas *et al.* , 2007 and see chapter 2) will be rewilded and managed for other activities linked with wilderness. Such approach can be seen as land-sparing at the local scale (cultivated vs wild), while at a broader scale food production and wilderness will share the land (Merckx & Pereira, in review; Phalan *et al.* , 2011).

When a transition from "species conservation" to "species management" occurs, adapted policy tools will be needed (Henle *et al.* , 2013). Some of the species benefiting from rewilding and showing positive population trends since land abandonment begun are large mammals, which are often associated with human/wildlife conflicts (see chapter 2). If those populations were to increase substantially, it could be difficult to segregate them entirely to wilderness areas and mechanisms will have to be designed to allow for mitigation, compensation and/or cohabitation (e.g. large carnivores, in Boitani & Linnell, in press). The set of policy instruments that can address human/wildlife conflicts are: regulatory (i.e. referring to the management and control of species); economic (e.g. compensations for damages caused by wildlife, subsidies for technical development for the prevention of damages); and educational, directed at the civil society (Similä *et al.* , 2013).

Promoting rewilding to manage abandoned farmland means shifting the policies towards an ecosystem process-based conservation, rather than the static conservation of a set of species and habitats which is currently in use (Hochkirch *et al.* , 2013). Assisted restoration can be needed in the early stages of conservation, depending on the ecological filters that could prevent and/or limit the return to self-sustaining ecosystems (Rey Benayas & Bullock, in press and see chapter 4 and chapter 2). In particular, the restoration of disturbance regimes to rewild opened landscapes following the abandonment of pastoral activities will mean the need of wild, or semi-wild grazers (see chapter 4), which could be (re)introduced if no local population was present. Though the introduction of wild species is legally framed (IUCN, 2013b), it is not the case for domestic species, such as horses, which could be used to maintain the disturbance regime of abandoned pastures. This calls for a legal framework on their reintroductions and on the liability of the various stakeholders involved (Jones-Walters & Čivić, 2010).

Rewilding will help policy-makers and stakeholders in rethinking their relationship with nature. In particular, the opportunity given by farmland abandonment to passively restore millions of hectares of land could give Europe an occasion to end the trends of double-standards between developed and developing countries in regard to conservation policies (Meijaard & Sheil, 2011). In order to do so, rewilding needs to gain visibility in the public and political sphere, as saliency (e.g. mainstreaming the concept of rewilding) has proven to be essential to the integration of concepts and ideas into the policy agendas (Jørgensen *et al.* , 2014; Rudd, 2011). In particular, rewilding research should aim at having three important impacts on policy makers (Rudd, 2011): a conceptual impact (to change the way policy makers think), an instrumental impact (to directly influence existing policies and managements), and a symbolic impact (to support established positions).

Changes in what societies want to preserve, and how they protect it have already been observed (e.g. Pinto & Partidário, 2012). The designation and management of PAs in Europe has evolved since the 1970s (Fig. 6.1), giving increasing importance to the role of local communities in managing the areas, and to the benefits that they should get from those (Jones-Walters & Čivić, 2013). For better or for worse, throughout decades of transitions in the way biodiversity is preserved, conservation baselines shifted, decision makers and stakeholders adapted, and so did the management approaches. Bringing rewilding in the agenda of conservation policies by showing its potential to both tackle the issue of land abandonment and restore wilderness could lead the way to a new transition of biodiversity conservation in Europe.

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7. Synthesis and future research avenues

"Nature is ever shaping new forms: what is, has never yet been;
what has been, comes not again."
Johann Wolfgang von Goethe (1783)

Synthesis and future research avenues

The aim of this thesis was to evaluate the potential of rewilding as a restoration opportunity, particularly to tackle the issue of land abandonment in Europe.

The following section synthesizes the main findings of the thesis in order to answer the questions asked in the introduction, to fill some knowledge gaps on rewilding and to pave the way for its adoption as a restoration and land management option in Europe. Further perspectives for research on rewilding are also discussed in each of the sub-sections.

Defining rewilding in a European context

The work presented in this thesis contributes to the definition of the concept of rewilding, applied to the European context.

Rewilding is the restoration, or land management plan, best applied for the transition from an “abandoned land” to a “wild land” (Fig. 7.1). For as long as the demand for agricultural goods, and the productivity of the soils are sufficient, “cultivated lands” can remain in this state. Yet, in some cases, farmland abandonment, enforced by socio-economic drivers, can occur and lead to “abandoned lands”, unless external factors, such as subsidies can inhibit the phenomena. If passive restoration is chosen, rewilding is the next step to allow the passive restoration from “abandoned lands” to “wild lands”. Yet, the existence of ecological filters can prevent/delay the rewilding processes, until assisted restoration is applied on the land. An option that has yet to be implemented is the development of policies in order to redirect subsidies for the maintenance of non-productive agriculture towards the facilitation of rewilding. Once rewilding occurred, the self-sustainability of the systems allows them to remain in a “wild state”.

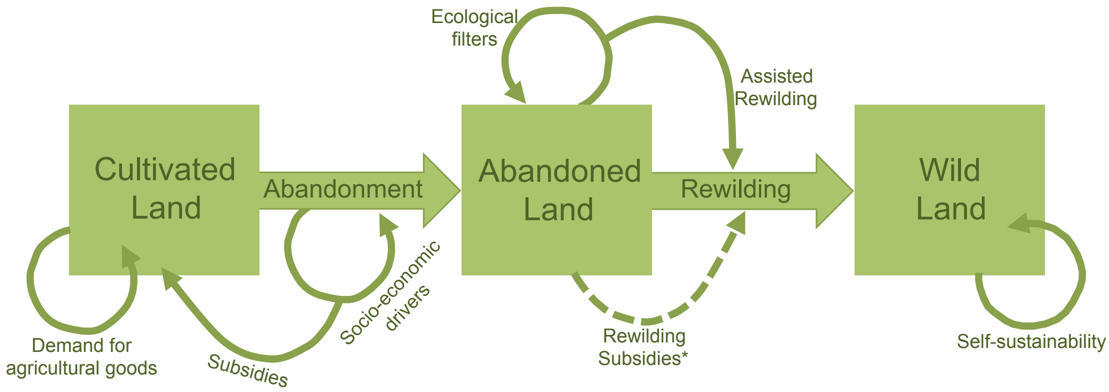

Figure 7.1.: Conceptual diagram placing rewilding, in a continuum of land transitions, from cultivated lands to wild lands. See details in the text. (*Rewilding subsidies do not yet exist)

The "re" of "rewilding" is sometimes interpreted as a return to a static and past state of the system, for example pre-agriculture (e.g. Boitani & Linnell, in press). However, though the study of pre-Holocene ecosystems can give guidelines to restore lost functions in the abandoned systems (Jackson & Hobbs, 2009; Sandom *et al.* , 2013a), the "re" merely means a return to a "wild" state, i.e. a state that is not dependent on human management. In this sense, choosing a functional baseline for rewilding is to be preferred to an historical baseline.

Current and projected extent of farmland abandonment and rewilding in Europe

In this thesis, we contribute to identifying the potential for rewilding in Europe, by estimating that the amount of land that could be converted from agricultural to natural by 2030 could vary between 9.9 and 29.7 Mha.

Current and predicted farmland abandonment is associated with less productive and remote areas of Europe. We identify the hotspots of rewilding in mountainous and marginal regions of the Alps, the Apennines, Northern Portugal, but also in Northwestern France, and Central Europe.

A return to a rural lifestyle is sometimes argued in response to the EU economic crisis but we hypothesize that such migration would be done towards peri-urban areas rather than remote and less productive places such as the ones currently facing abandonment.

For the research presented in this thesis, we used a land-use change model capable of predicting land abandonment (Verburg & Overmars, 2009). Nonetheless, rewilding is not explicitly considered as a land management option in land-use change models. Further research in the area would involve the design of scenarios where European policies consider rewilding as a possible management in order to better assess its potential for land restoration, biodiversity and ecosystem services.

Understanding the interactions between societies and the land-use.

Land abandonment creates an opportunity for an increase in wild areas, which could be restored by rewilding. However, such opportunity will depend on the dynamics of the rural populations in these areas.

We contribute to the understanding of these dynamics by analyzing a socio-ecological model applied to a particular hotspot of rewilding, in Northern Portugal. We use this theoretical approach to analyze a very specific case of seasonal migration in the area of Peneda Gerês, providing a first insight in the understanding of the dynamic between the nomad/sedentary populations and the forested area linked with the summer and winter villages.

Our results show that the coupled dynamic exhibits the existence of tipping points in the systems. These findings are particularly well suited to cases where a tight community will tend to adopt the same behavior (e.g. sedentarization), further enforced by encroachment and forest regeneration, thus making a return to the initial state (e.g. cultivated land) unlikely, and allowing the forested area to expand where sedentary residents abandoned the land.

We show that developing socio-ecological models and identifying the tipping points within the systems is of particular importance to make accurate scenarios of change (Leadley *et al.* , 2010; Walker & Meyers, 2004), including land use change such as farmland abandonment and rewilding.

Future research in this area should first focus on calibrating and validating the models with empirical data. Another research avenue could focus on the scaling-up of socio-ecological models, for example from a local to a national dynamic. Also, the implementation of spatially explicit versions of the model would improve the relevance of the outputs for researchers, land planners and decision makers.

Assisted rewilding can overcome ecological filters to passive restoration

Though most damaged ecosystem can revover from human impact (Jones & Schmitz, 2009), their resilience is dependent on the existence, or not, of ecological filters that can limit natural successions and rewilding.

Assisted passive restoration can facilitate and/or accelerate the process of rewilding when ecological filters would hamper it (Pereira *et al.* , subm; Shono *et al.* , 2007). In particular, the work presented in this thesis suggests to restore the regimes of natural disturbances that were replaced by humans activities which are now abandoned, in order to maintain an heterogeneous, yet wild, landscape. In particular, we propose the use of prescribed burning and the reinforcement or reintroductions of populations of wild herbivores, in order to trigger the disturbance regimes and maintain a mosaic of habitats.

Nonetheless, these assumptions have yet to be tested in a rewilding context. In particular, the interest of "assisted" rather than "active" restoration is the short-term management implication until the systems become self-sustaining. Further empirical studies on both the ability of secondary successions to occur on abandoned land, and the efficiency of assisted rewilding should be conducted. Coupled models of disturbance regimes and land-covers dynamics should also be developed in order to determine which are the necessary conditions for rewilding to occur passively, or for efficient assisted restoration.

Rewilding can be an opportunity for biodiversity

Most studies performed on the consequences of agricultural abandonment in Europe, and rewilding, for biodiversity tend to show negative impacts (Queiroz *et al.* , 2014). Nonetheless, this thesis illustrates that several species, including some that were not so long ago locally extinct in several countries, do benefit from both a decreased human pressure, and a higher availability of the land.

An argument often used to link biodiversity richness and human activities is the "intermediate disturbance hypotheses" (e.g. EEA, 2004). In this context, richness will peak at a non-null level of disturbance, which has been interpreted as a positive impact of low intensity agriculture for biodiversity. Indeed, traditional and extensive agriculture creates and maintains a mosaic of habitats and allows for heterogeneous landscapes where several species can thrive (e.g. Blondel, 2006). Yet, this interpretation of the intermediate disturbance hypothesis disregards several aspects of the disturbance/biodiversity relationship. First, it disregards natural disturbances and their potential in creating and maintaining a heterogeneous landscape. Second, the (positive) impact of anthropogenic disturbances will depend on the scale at which they are investigated (e.g. Merckx, in press), and on the metric of biodiversity that is used. To further investigate the intermediate disturbance hypothesis, the response of biodiversity to low intensity farmland, abandonment, and rewilding should be investigated across scales and across taxa.

Rewilding and wild areas can improve human well being

In this thesis, we present the first qualitative assessment (at the EU scale) of the link between the supply of ecosystem services and the wilderness quality index, and the first quantitative analysis of the potential benefits of rewilding (at the Iberian scale).

We identify the potential contribution of rewilding to human well being through an increase in the supply of ecosystem services. In particular, the supply of regulating (e.g. carbon sequestration, water purification) and cultural services would be improved by an increase in the wild areas resulting from agricultural land abandonment and rewilding.

To further improve the knowledge on the potential of rewilding and wild lands for the supply of ecosystem services, the same type of quantitative analysis should be expended at the EU scale, though in this case, an assessment of the average supply of each service at the eco-region level would be more relevant.

Identifying which ecosystem services would have an increased supply and where will also contribute to the inclusion of rewilding in the EU conservation policy agenda.

There is room for rewilding in the EU conservation policies

A framework for rewilding in the EU policy does not yet exist but the work presented in this thesis exposes some ecological and economical arguments that can contribute to accelerating the inclusion of rewilding in the conservation agenda of Europe.

Indeed, rewilding currently contrasts with the land conservation policies that are in place in Europe, where active management is emphasized by maintaining, or mimicking, low-impact extensive agriculture in certain areas. Yet, the recent reform of the Common Agricultural Policy for the 2014-2020 period does not seem to deliver on the "greening" expectations that were placed on it (e.g. Pe'er *et al.* , 2014), while we show that rewilding can also prove to be efficient to maintain those heterogeneous landscapes.

Furthermore, as argued in this thesis, implementing rewilding in Europe can lead to an increased area of wilderness, and contribute to the EU and global targets set for 2020, both in terms of biodiversity conservation and ecosystem services supply.

One of the major remaining limitations to the implementation of rewilding in Europe is the negative perception of farmland abandonment and rewilding, particularly the tolerance that the public can have for a wildlife comeback (Enserink & Vogel, 2006). This is mainly the case when it is accompanied by physical (e.g. damage to crops, depredation on livestock) and/or psychological (fear of attacks, "bad reputation" of a species) conflicts with local communities, and little access and information on mitigation and compensation measures (Treves & Bruskotter, 2014). Hence, in order to facilitate the adoption of rewilding by the various stakeholders and the EU decision makers, better information and education, combined with efficient mitigation and compensation schemes should be investigated and implemented.

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A. Appendix: Pereira et al. 2012

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In this publication, Laetitia M. Navarro participated in the building up of the database on species distribution, IUCN category and threat, based on IUCN[1] and BirdLife[2] data. This database was used for the elaboration of figures 3, 5, 6, 7 a,b, and c, and figure 8. LMN also produced figures 1, 2, 4 c and d, 7 b and d, and Table 1. Additionnally, LMN contributed to the text, particularly in section 7: ”A bit of good news for a change”.

[1]IUCN (2011). IUCN Red List of Threatened Species. Version 2011.2.
[2]BirdLife Int. (2011). Distribution Maps of Birds of the World. Cambridge, UK.

Global Biodiversity Change: The Bad, the Good, and the Unknown

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Abstract

Global biodiversity change is one of the most pressing environmental issues of our time. Here, we review current scientific knowledge on global biodiversity change and identify the main knowledge gaps. We discuss two components of biodiversity change—biodiversity alterations and biodiversity loss—across four dimensions of biodiversity: species extinctions, species abundances, species distributions, and genetic diversity. We briefly review the impacts that modern humans and their ancestors have had on biodiversity and discuss the recent declines and alterations in biodiversity. We analyze the direct pressures on biodiversity change: habitat change, overexploitation, exotic species, pollution, and climate change. We discuss the underlying causes, such as demographic growth and resource use, and review existing scenario projections. We identify successes and impending opportunities in biodiversity policy and management, and highlight gaps in biodiversity monitoring and models. Finally, we discuss how the ecosystem services framework can be used to identify undesirable biodiversity change and allocate conservation efforts.

Contents

1. INTRODUCTION

Biodiversity is the sum of all "plants, animals, fungi, and microorganisms on Earth, their genotypic and phenotypic variation, and the communities and ecosystems of which they are a part" (1, p. 138), or simply stated, life on Earth (2). Biodiversity is multidimensional, and no single measure of biodiversity can capture all its dimensions (3). Biodiversity provides the foundation for ecosystem services, including nutrient cycling, climate regulation, food production, and the regulation of the water cycle, and it is therefore intimately linked with human well-being (2, 4, 5). This foundation is now becoming endangered as the human footprint on the planet increases and biodiversity declines. Species are becoming extinct at rates higher than in the fossil record of the past few million years, including the peak extinction rate owing to the megafauna disappearance at the end of the Pleistocene (6). Several other dimensions of biodiversity are also declining, such as the extent of tropical forests and the mean abundance of wild bird species (7, 8). The human appropriation of Earth's natural resources is not only leading to biodiversity loss but also to large alterations of biodiversity distribution, composition, and abundance.

Here, we review our current understanding of global biodiversity change and its underlying drivers. We start by scoping our definition of global biodiversity change, which includes both biodiversity loss and biodiversity alterations. Next, we briefly review human-induced global biodiversity change since the last ice age to the Industrial Revolution. This provides a historical background for our discussion of recent biodiversity change, which is organized into four biodiversity dimensions: species extinctions, species abundances and community structure, species ranges, and genetic diversity. These dimensions are not by any means exhaustive but aim at being representative. We focus on terrestrial ecosystems, but we also give examples for freshwater and marine ecosystems. Next, we examine the direct drivers of biodiversity change: habitat change, overexploitation, pollution, biotic exchange, and climate change. Some of these drivers could also be considered dimensions of biodiversity, such as the change in quality of a habitat or biotic exchanges, but for simplicity, we treat them only in the drivers section. We discuss how these drivers might evolve in the next few decades by reviewing existing social-ecological scenarios and the projections for indirect drivers, such as population growth, consumption patterns, and energy use. Although much of the news related to biodiversity change is worrying, we also provide an overview of future opportunities for reversing biodiversity declines and increasing biodiversity at the local level, as well as review some recent successes in biodiversity conservation. The next section discusses the gaps in our understanding of global biodiversity change, both in observations and modeling. We conclude with some thoughts on the nature of biodiversity change and the need to focus our management efforts on detrimental biodiversity change.

2. GLOBAL BIODIVERSITY CHANGE: ALTERATIONS AND LOSSES

Many organisms modify the environment and as a result increase their fitness or affect resource availability to other species, processes known as niche construction of ecosystem engineering (9). Humans and their hominid ancestors are no exception; they have been modifying ecosystems throughout history to improve food availability and decrease the success of their ecological competitors. What is truly exceptional about humans is the scale at which they have been able to modify ecosystems. The total industrial fixation of nitrogen (mainly for fertilizer production) together with biological fixation in crops, and nitrogen mobilized during fossil-fuel combustion, is greater than the nitrogen fixed by all natural processes together (10). Humans currently harvest about 15% of global terrestrial net primary production, using about six times more net primary production than was used by the extinct Pleistocene community of megaherbivores (11). More than 35–40% of the world's forests and other natural ice-free habitats have been converted to cropland and pasture (12, 13), a value that increases to about 70% in some biomes, such as Mediterranean forests (2). Over half of the world's large river systems have been affected by dams (14), and 40% of the ocean is strongly affected by multiple drivers (15). Some of these impacts do not target specific species, such as altering the nitrogen cycle or land-use change, but may favor some functional groups. Other actions are directed at specific species or at least aim directly at some functional groups, such as hunting, fishing, and timber logging.

An important distinction should be made between biodiversity loss and biodiversity alterations (**Figure 1**). This issue is particularly important as it implies that not all biodiversity change is inherently a bad thing, and therefore we often need to define a set of criteria to assess the benefits and disadvantages of biodiversity change. Recent global species extinctions correspond to net biodiversity loss, as the number of species created by evolutionary processes occurs at a much slower pace than the recent extinction rates (6, 16). The loss of genetic diversity, particularly the disappearance of populations and particular alleles, also corresponds to biodiversity loss, although small alterations of genetic diversity may not correspond to significant biodiversity loss.

Much of human action alters the species composition and the relative species abundances in an ecosystem, changing the structure

Biodiversity: the sum of all organisms on Earth, their variation, and the ecosystems of which they are a part

Biodiversity loss: the local or global extinction of an allele or species

Drivers: direct or indirect pressures on biodiversity that induce a change (either negative or positive)

Biodiversity alterations: human-induced changes that lead to modifications of community structure or to shifts in species distributions

Scenarios: plausible stories about how the future may unfold, often associated with quantitative projections

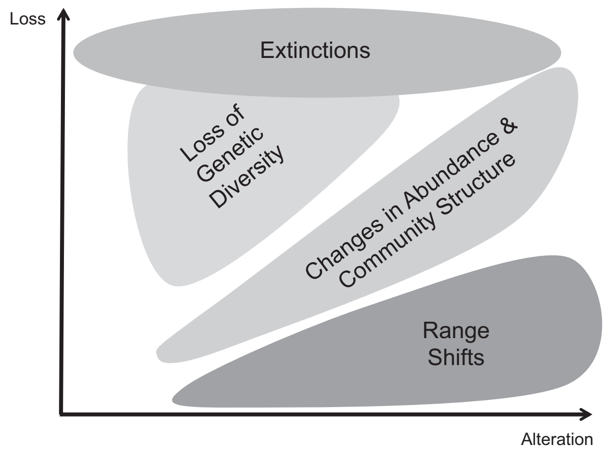

Figure 1

Conceptual diagram illustrating the intensity of loss and alterations associated with the different dimensions of biodiversity change: extinctions, loss of genetic diversity, changes in abundance and community structure, and range shifts.

of communities, but may not lead to biodiversity loss at the regional or global scale (**Figure 1**). For instance, the conversion of farmland into forest may lead to the decline of farmland bird populations but result in a population increase of forest species (17, 18). Still, large alterations in abundance and trophic structure may cause net biodiversity loss (**Figure 1**). For instance, the depletion of fisheries (19) or the overall decrease in the Living Planet Index (20) can certainly be considered net biodiversity loss.

Many shifts in species' range induced by climate or abiotic factors may not lead to a net biodiversity loss at the global scale (**Figure 1**). However, a local scale analysis can produce a very different result. Shifts in species distributions occur when a species goes locally extinct in some parts of its former range and colonizes new sites. Therefore, in a place where the species goes extinct, one can consider that biodiversity has been lost, while in a place that a species has colonized, one can consider that biodiversity has been gained. This last interpretation is however context dependent: The expansion of exotic species leads to an overall homogenization of global biodiversity that is arguably making the biosphere more monotonous and can threaten native species. The rearrangement of communities may also lead to the development of new communities, particularly for regions where new climates without current analogs develop (21).

3. A BRIEF HISTORICAL PERSPECTIVE ON GLOBAL BIODIVERSITY CHANGE: FROM THE ICE AGE TO THE INDUSTRIAL REVOLUTION

We can hypothesize that the first actions of humans with large-scale impacts on biodiversity were fire and hunting. It is difficult to date precisely when humans started controlling and manipulating fires. There have always been natural fires associated with lightning and volcanic activity, and therefore the co-occurrence in an archeological site of burning and artifacts does not necessarily imply a causal link between the two (22, 23). The first intentional uses of fire were likely domestic, including cooking, heating, predator defense, illumination, and artifact manufacture and may have started as long as 1.9 Mya ago, although its widespread use seems to date back only to the beginning of the Middle Paleolithic, around 400,000–200,000 years ago or even later (**Figure 2**) (23–25). However, the systematic use of fire as an ecosystem management tool is perhaps much more recent, beginning tens of thousands of years ago (24). Landscape burning has several purposes, which include driving game into hunting areas, clearing thick vegetation for travel, and opening up grazing areas for game species (26). We know that some recent hunter-gatherer societies, such as Native America tribes and Australian Aborigines, managed landscapes with fire and that fire also played an important role in early agrarian and herding societies to maintain open vegetation and fertilize soil (26). Identifying how early landscape management by fire became a tool in hominids is harder, and a recent study has not found a significant difference in fire regime between the Neanderthal occupation and the arrival of modern humans in Europe (26). Evidence for change in fire regime in Southeast Asia and Australia goes back to about 40,000 years ago, but the Australia evidence has faced some recent challenges (26).

Hunting is likely to have driven the first wave of species extinctions induced by humans starting 50,000 years BP (**Figure 2**) (22, 27). The extinction of large-bodied vertebrates (i.e., megafauna; >44 kg) closely followed the global spread of *Homo sapiens* to new continents and islands. In Australia, 88% of the megafauna mammal genera went extinct between the time of human arrival, ≈50,000 years BP, and 32,000 years BP (28). In North America, 72% of the megafauna mammal genera went extinct, mostly between 13,500 and 11,500 years BP (28), and shortly after the arrival of humans in the continent between 15,000 years BP (29, 30) and 13,000 years BP (31). In South America, 82% of the genera went extinct sometime between 12,000 and 8,000 years BP

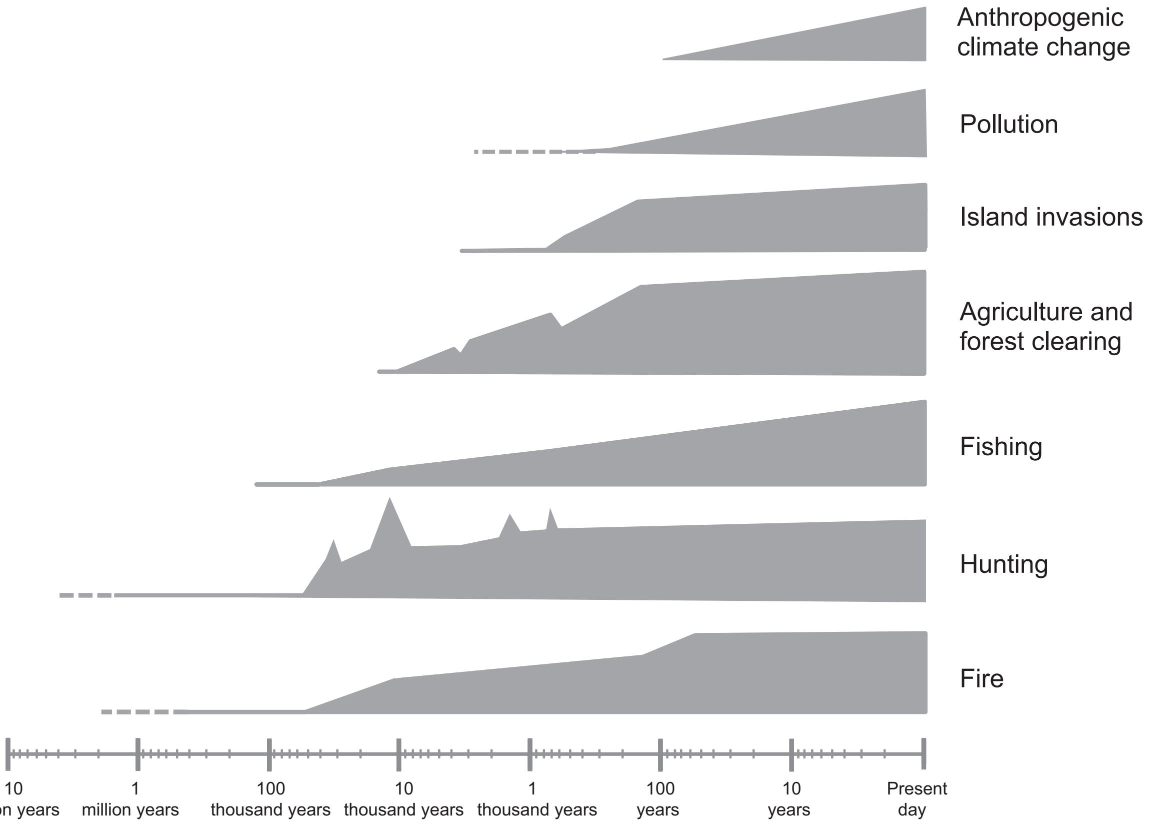

Figure 2

Qualitative representation of the temporal evolution of the main anthropogenic drivers of biodiversity change. References used for dating the pressure trend of each driver: fire (23, 24), hunting (28), fishing (160), agriculture and forest clearing (36, 40, 41), species invasions on islands (42), pollution (2), and anthropogenic climate change (138).

(28). Megafauna also went extinct in the large islands of Madagascar (e.g., giant lemurs) and New Zealand (e.g., ten species of moa) soon after human arrival at about 2,000 years and 1,000 years ago, respectively (22, 27). The relative roles of human hunting versus climatic changes in driving megafauna extinctions have been hotly debated (32), but it is now becoming accepted that, although climate may have contributed to preempt the conditions for the megafauna decline, humans played a major role in accelerating extinctions through hunting (27, 28). The megafauna extinction had major impacts in ecosystems, including on the fire regime, seed dispersal regime, and ecosystem function and structure (33, 34).

The next large-scale impact on ecosystems came with the development of agriculture (**Figure 2**) (35). There were multiple origins of crop domestication: einkorn wheat, emmer wheat, barley, rye, lentil, pea, bitter vetch, chickpea, and flax, starting about 10,000 years BP in the Fertile Crescent (36); rice, soybean, and foxtail millet in East Asia at about the same time (37); and squash, peanut, quinoa, and cotton between 9,000 years and 6,000 years BP in parts of the Andes (38). Agriculture rapidly radiated from these regions to other regions occupied by humans, although at a faster rate in Eurasia than in the Americas or sub-Saharan Africa (39). But agriculture was not only the domestication of crops. Domestication of animals was a key component of the development of agriculture, particularly in the Fertile Crescent, where sheep, goats, and pigs started being domesticated around the same time as the plants (40). Over millennia, agriculture would bring major ecosystem changes with

IUCN: International Union for Conservation of Nature

the deforestation of large areas, changes in fire regime, the appropriation of primary productivity by humans, and the replacement of wild herbivores by domestic grazers (11, 28, 41). In Europe, by 3,000 years BP, perhaps as much as 30% of the usable land for crops and pasture had already been cleared (41), a pattern that would continue to intensify over the following centuries, only briefly interrupted by the Dark Ages (AD 500–700) and the black death (AD 1350). At the beginning of the Industrial Revolution, at around AD 1850, the usable land cleared for agriculture in Europe may have reached a peak of about 80% (41), much higher than what is currently observed.

The most recent wave of extinctions before the Industrial Revolution occurred in islands and was likely associated with the expansion of global trade via maritime routes (**Figure 2**). Between AD 1500 and 1800, all documented extinctions occurred on islands (42). Bird extinctions are particularly well documented for that period. The major drivers of bird extinctions have been, by decreasing order of importance, invasive species, overexploitation, and habitat loss (42). The effects of invasive species, such as cats, rats, and goats, included both direct predation upon the native birds or the degradation of their habitats (43).

4. RECENT TRENDS IN GLOBAL BIODIVERSITY CHANGE

In this section, we review what is known about global biodiversity change since the Industrial Revolution (mid-nineteenth century onward). Much of the emphasis is on very recent changes in the past 40 years, as some of the data are only available for that period. We divide our analysis into four different dimensions of biodiversity change that have different scores in the loss and alteration axes (**Figure 1**).

4.1. Species Extinctions and Extinction Risk

During the twentieth century, there were approximately 100 extinctions of birds, mammals, and amphibians (16). Considering that there are approximately 21,000 species described in these groups, this yields a rate of 48 extinctions per million species years (E/MSY), about 20 to 40 times greater than the average extinction rate for the Cenozoic fossil record of 1–2 E/MSY (6). Unfortunately, much less is known for other taxonomic groups and for organisms inhabiting the marine (44) and freshwater realms (45). In the very recent period of 1984–2004, the International Union for Conservation of Nature (IUCN) recorded 27 extinctions (42). Approximately half of these extinctions have occurred on continents, suggesting that recent extinctions are no longer mostly restricted to oceanic islands. Twelve of the extinct species were flowering plants, followed by eight amphibians and six bird species. Habitat loss is thought to have played a role in 13 of these extinctions, followed by invasive exotics and disease (particularly the amphibian disease chytridiomycosis). Habitat loss seems therefore to be playing a much larger role in very recent extinctions than in previous centuries, and disease is emerging as a new threat (42).

The current importance of habitat loss and degradation is also apparent from analysis of the IUCN Red List of Threatened Species (**Figure 3**) (42, 43), where it is identified as the main current threat to amphibians, mammals, and birds. The Red List identifies not only the species that have been confirmed to have gone extinct but also the species that are currently threatened and, if pressures remain, may become extinct in the future. This allows for a more immediate analysis of global biodiversity change, as the lag between the initial decline resulting from a pressure, such as habitat loss, and the final extinction may take centuries or millennia (46, 47). Furthermore, a species may become functionally extinct with a major impact on ecosystem processes and services much before it becomes extinct in the wild: Examples include the disappearance of birds playing a major role in seed dispersal and pollination (48) and the collapse of fisheries (19, 49). The Red

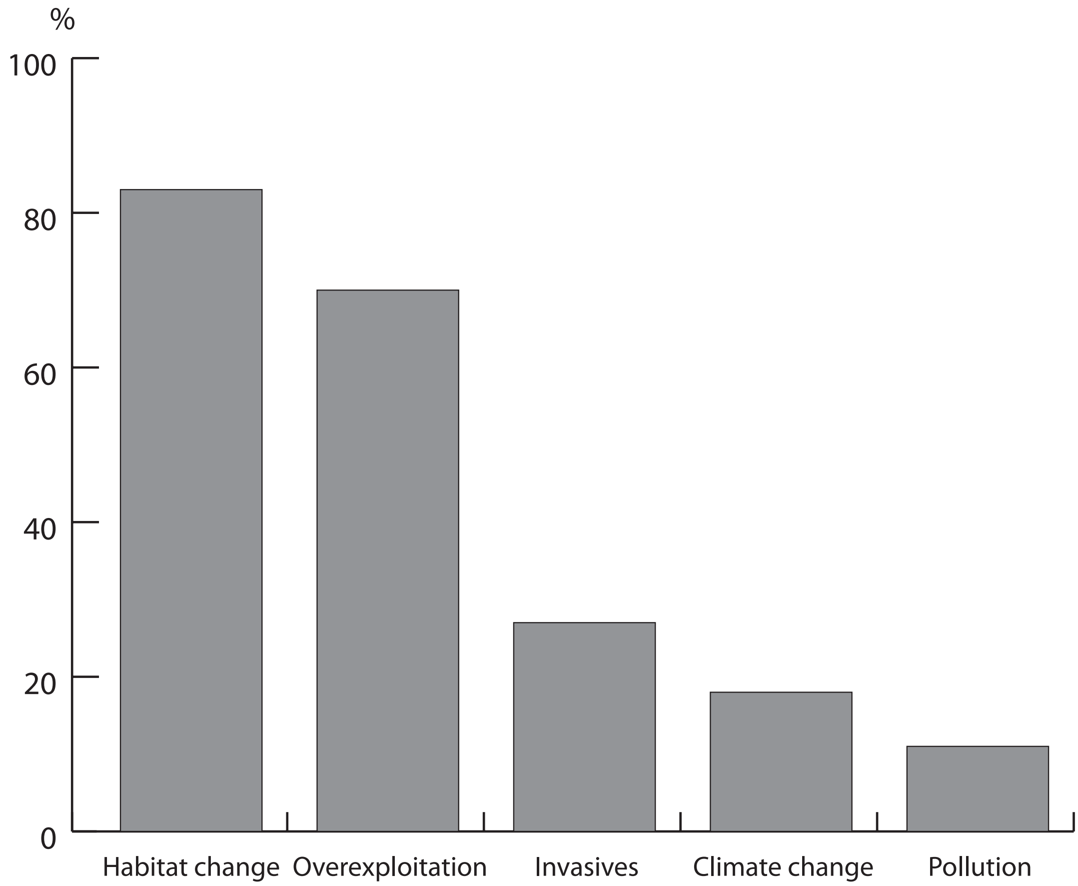

Figure 3

Proportion of threatened species affected by each driver. Threatened species ($n = 4,259$) include mammals, birds, and amphibians in the following Red List categories: critically endangered, endangered, and vulnerable. Main threats are classified as habitat change (i.e., residential and commercial development, agriculture and aquaculture, energy production and mining, transportation and service corridors, and natural system modifications), overexploitation, invasive species, climate change, and pollution (161). Several threatened species are affected by multiple threats.

List uses objective criteria to assess the degree of threat to a species into one of seven major categories of increasing risk (50): least concern, near threatened, vulnerable, endangered, critically endangered, extinct in the wild, and extinct. Species that have been assessed by the IUCN but for which insufficient data are available to define the threat category receive a data-deficient classification. Of the 30,738 species from taxa representatively assessed in 2010, 23% were threatened (**Figure 4*a***), assuming that the proportion of threatened species for data-deficient species is the same for data-sufficient species. This is a high proportion and reflects the seriousness of the biodiversity crisis. Nonetheless, it must be interpreted with care because the approach used to assess threat includes not only population and geographic range reductions (extrinsic factors), but also characteristics of the species, such as small population size and restricted geographic range (intrinsic factors). Species may exhibit these characteristics naturally, and they may not be correlated with human-induced extinction risk (51).

Another problem is the taxonomic bias of the assessed species. We still do not know how many species exist on Earth, with a recent estimate placing the total number of species at 7.4 to 10 million (52). Of these, only around 1.7 million species have been described (**Figure 4*a***) (16, 43). Systematic global Red List assessments have been carried out for only a few taxonomic groups, and the proportion of species assessed in each group is very different from its representation in global biodiversity (**Figure 4*a***). It is virtually impossible to assess the extinction risk of all taxa. Instead, in the past few years, the IUCN has developed a randomized sampling approach to expand its assessment to more taxonomic groups (53).

Still, the overall pattern emerging from the Red List assessments is that amphibians (41% threatened) and cycads (63% threatened) are the most threatened groups, and birds are the least threatened group (13% threatened) (54). The generally low mobility and small ranges of amphibians and cycads may contribute to this vulnerability, but one might also ask if our better knowledge of bird species has contributed to their lower assessment of threat.

Most of the threatened terrestrial vertebrates occur in tropical regions (**Figure 5*b***), following the latitudinal trends in the species richness of this group (**Figure 5*a***). A very different map is obtained by looking at the relative proportion of threatened species in each grid cell (**Figure 5*c***). Incidence of threatened species is high in much of Asia (except the North), the Sahara, the Andes, Madagascar, the Caribbean, New Zealand, and other islands. Areas of high species diversity and moderate to high incidence of threatened species include the Indo-Malayan region (particularly Southeast Asia), the Andes, Central America, the Brazilian Cerrado and Atlantic Forest, and some localized areas of sub-Saharan Africa (**Figure 6**). These are regions with restricted-range species (42, 55), and most have undergone rapid forest loss (56, 57).

The pattern for threatened marine vertebrates (cartilaginous fish) is somewhat similar, with higher occurrence of threatened species in the tropics, but there is also a strong coastal signal, with both of these regions having higher species richness (54). When one controls for the species richness effect, high incidence of threatened species is still found at coastal areas (54). This pattern agrees with the higher human pressure on coastal regions, particularly that associated with fishing activities (15).

The Red List status gives us a snapshot of what is happening to biodiversity at a given time. However, we are also interested in understanding the trends in biodiversity. The Red List Index compares the proportion of species in the different threat categories over time (43, 54, 58). A key component of developing the Red List Index is the identification of species that have changed status not because more information became available but because the conservation situation of the species changed, i.e., genuine changes (43). Red List Indices have been calculated for birds (1988–2008), mammals (1996–2008), amphibians (1996–2008), and corals (1996–2008) (8, 54, 59). In all cases, the Red List Index shows an increase in the proportion of threatened species, and this increase is especially pronounced for corals owing to the large-scale bleaching event of 1996–1998. It is important to understand that a flat (or unchanging) Red List Index means, in theory, that species are still declining toward extinction, as the maintenance of a given category of threat indicates that a species population size or geographic range continues to decline at the same rate (60). This contrasts with the mean population abundance indices discussed in the next section, where a constant value means the maintenance of the relative extinction risk. However, the fact that the risk assessment includes both population/range size and population/range change can blur this distinction.

4.2. Changes in Species Abundances and Community Structure

Changes in extinction risk status can be slow and do not capture important alterations of ecosystem function that can occur when species abundances change (61, 62). In the past decade, several indicators have been developed to assess the population abundance dimension of biodiversity (**Figure 4*b***), including the Living Planet Index (LPI) (20, 63), the European Common Farmland Bird Indicator (64), the Wild Bird Index (WBI) (covering North America and Europe) (8), and the European Butterfly Indicator for Grassland Species (65). Most of the data in these indicators comes from extensive observation networks of volunteers (66), and they portray one of the most immediate and detailed pictures of global biodiversity change. The idea in each of the indicators is to obtain the average population trend across a set of species and their populations. For example, the LPI includes 7,190 vertebrate populations from 2,301 species across the marine, freshwater, and terrestrial realms (8). The LPI for a given year is based on the geometric mean across all populations of the relative abundance indices between that year and the previous year (i.e., $\frac{N_t}{N_{t-1}}$). That geometric mean is then multiplied by the value of the LPI in the previous year to give the final index value for that year, starting in 1970 with the value 1 (or 100%). The geometric mean of relative abundance indices has nice statistical properties, particularly when based on common species, and is able to detect overall abundance and evenness decreases and, to some extent, species richness decreases (67, 68). The geometric mean is equal to one when the halving of the density of a species is compensated by a doubling of the density of another species. The other indicators mentioned above follow similar approaches.

Overall, the pattern that emerges from all of these indicators is of global or regional declines of species abundances, despite some year-to-year fluctuations of some indicators (**Figure 4*b***). The LPI has declined from 1970 to 2006 by 31% (8), the WBI for habitat specialists has declined from 1980 to 2007 by 2.6% (8), the European Common Farmland Bird Indicator declined from 1980 to 2006 by 49% (69), and the European Butterfly Indicator for Grassland Species declined from 1990 to 2009 by 70% (based on a best-fit line) (65). These numbers paint a depressing figure and are in some cases large enough to suggest that ecosystem processes and services are being modified (70, 71). However, they must be interpreted with some caution as the spatial and taxonomic coverage of these indicators is limited (72). Furthermore, a finer analysis of these indicators can tell some contrasting stories. The tropical terrestrial LPI has declined, but the temperate terrestrial LPI has increased (20). One possible explanation is that, although tropical ecosystems are now undergoing fast and detrimental land-use change and overexploitation (2), these drivers peaked in temperate regions much before 1970 and are now decreasing as a consequence of farmland abandonment, greater species protection, and conservation actions. This has favored the return of large mammals (those that survived the earlier extinction wave) and other species in some temperate regions (17, 73). Still, the same habitat changes that have benefited large mammals are thought to contribute to the decline of farmland birds and grassland butterflies, although agricultural intensification is likely to play a major role too (64, 65). There are major geographic differences in the marine LPI, with strong decreases in the Indian Ocean and Southern Ocean and increases elsewhere (20). Similarly, although the terrestrial species in the Wild Bird Index have declined by 16%, the wetland species have increased by 40%. This last case also illustrates the problem of spatial coverage: The Waterbird Population Status Index, with global coverage and measuring the proportion of monitored shorebird populations, declined 18% from 1985 to 2005 (8). We discuss biodiversity change uncertainties associated with spatial coverage in detail in Section 6.

In the marine realm, much of the existing data come from fisheries, which have influenced the development of marine biodiversity indicators. The Marine Trophic Index (MTI) measures the mean trophic level of fish landings (74). The MTI declined globally in the 1960s and in the 1980s, and it increased in the early 1970s and since the 1990s (8, 75). Declines have been attributed to overfishing of large species, leading to shifting fishing efforts to smaller species at lower trophic levels. Recent increases have been attributed to the spatial expansion of the fishing effort (8, 76). The sensitivity of the

LPI: Living Planet Index

Biodiversity indicator: a metric used to assess the rate and intensity of biodiversity change

MTI to changes in the spatial distribution of fishing effort has led to the search for alternative measures of species abundance changes in the oceans. One such measure is the proportion of fish stocks not fully exploited or depleted (**Figure 4*b***). For the fisheries that have been assessed, this proportion has decreased to half since 1974, and currently, only 21% of the stocks are not fully exploited or depleted (8, but see Reference 19 for an alternative estimate). Although the MTI and the proportion of fully exploited stocks give us a measure of the capacity of the ecosystem to provide a service, they may not reflect the overall state of biodiversity in those systems, as many species are not targeted by fishing.

Coastal habitats have been undergoing particularly high human pressure (77), and coral reefs, one of the most biologically diverse and productive systems on the planet, are particularly vulnerable because of their sensitivity to climate change and other pressures (78, 79). One measure of the community structure of coral reefs is hard-coral live cover (80, 81). Hard-coral live cover had a marked declined in the late 1970s in the Caribbean, following the white band disease outbreak, but has remained steady since the mid-1980s, although other community changes have been observed, including an increase in macroalgae cover in the late 1980s (**Figure 4*b***) (81). In the Indo-Pacific live hard-coral cover has declined since the 1980s, and particularly from 1997 to 2004 (80), and in 2003, coral cover averaged 22.1%, a value much lower than the historic baseline estimates of >50% cover. The bleaching event of 1996–1998 has had major impact, but disease, sedimentation from coastal development, and destructive fishing practices have also played a role.

4.3. Shifts in the Distribution of Species and Communities

Climate change and other ecosystem change drivers may cause alterations in species distributions (3, 82, 83). The alteration of a species distribution can be decomposed into two major aspects: directional shifts in the distribution (3) and changes in the size of the distribution (84). Directional shifts have been measured using species distribution centroids (3) or range limits (85). Recently, a new measure for directional shifts has been proposed, the Community Temperature Index, which tracks how the composition of communities at each site changes toward high-temperature dwelling species (86). Changes in the size of the species distribution are likely to be correlated with overall changes in species abundance (87, 88); however, directional shifts in the distribution may go undetected if only total species abundances are tracked.

An early meta-analysis of birds (United Kingdom), butterflies (Sweden), and alpine herbs (Switzerland) suggested that species were moving their range limits poleward at an average rate of 0.61 km/year (**Figure 4*c***) (85), providing evidence of climate change impacts on species distributions. Another study analyzing northern limit shifts across 16 taxonomic groups in the United Kingdom found average shifts of 1.2–2.5 km/year (**Figure 4*c***) (89). More recently, an assessment of distribution shifts for birds and butterflies in Europe, using the Community Temperature Index, has found rates of 2.1 and 6.3 km/year, respectively (**Figure 4*c***) (90). The one order of magnitude difference between the lowest estimate of range shifts and the highest estimate may be caused by the different methods used, the different regions, and the different taxa analyzed. The intervals of species shift rates are consistent with those of the velocities of isotherms from 1960 to 2009 in land surfaces (median of 2.7 km/year) and oceans (2.2 km/year), which exhibit large spatial variations, with some regions exhibiting no significant shifts and others shifting at rates higher than 10 km/year (91).

Average shifts may hide substantial variation in individual species responses, as some species maintain their previous ranges, others move toward the poles (i.e., North in the Northern Hemisphere), and yet others move in unexpected directions (83, 92). For instance, in the North Sea, the varying responses of different species (**Figure 4*c***) led to a nonsignificant change in the mean latitude of species ranges from 1980 to 2004, although most species assemblages tracked yearly fluctuations in climate with mean latitudinal shifts of 10–70 km/°C (83). Some species assemblages, such as warm-water specialists, exhibited significant overall shifts during these 25 years, moving northward at a rate of 4 km/year (83).

Species can also adapt to climate change by shifts in elevation (85, 92) or depth (83), shifting life history traits in time (93), or by adapting to the new conditions in their local range through phenotypic plasticity or microevolution (94).

4.4. Genetic Diversity in Domesticated and Wild Species

Of the four biodiversity dimensions analyzed here, we have the least information at the global level for changes in genetic diversity. Studies of loss of genetic diversity can be classified into two categories: studies of genetic diversity of domesticated species and studies of genetic diversity of wild species. Studies of domesticated species can further be divided into plant genetic resources (95) and animal genetic resources (96).

Over the past few decades, the worldwide adoption of modern crop varieties adapted to high-input systems has led to the reduction in the area farmed with local crop varieties (95). This change in agricultural practices has raised concerns: For instance in China, the number of rice breeds in production is reported to have declined since the 1950s from 46,000 to 1,000, and most of the 10,000 traditional corn breeds are no longer in production (97). Still, there are many farm communities that, although exposed to modern varieties, choose to maintain, at least in portions of the farm, traditional varieties (98). The picture of allelic diversity change is also complex. Although some studies report declines in allelic diversity of modern varieties over the past few decades (99), a meta-analysis of 44 studies has found no significant overall trend (100). Another concern is the status of the crops' wild relatives, which are under the same threats as other aspects of biodiversity, and recently a system of priority areas for their conservation in situ has been proposed (95).

Of the about 7,600 animal breeds (among 36 domesticated mammal and bird species) registered in the UN Food and Agriculture Organization's Global Databank, 20% are classified as being at risk, and a further 9% have become extinct (**Figure 4*d***) (96). Over the past decades, a similar phenomena to what happened with the crop varieties is occurring with the animal breeds: Local animal breeds are being replaced by widely used and high-output breeds more adapted to intensive animal production systems (7).

Less is known about the loss of genetic diversity in wild populations. One study has estimated that about 16 million populations are being lost annually, on the basis of an estimate of 220 populations per species derived from a review of population genetic studies and an assumption of linearity between tropical deforestation and population extinction rates (101). This is a very indirect estimate, and to our knowledge, it has not been confirmed independently. Other studies have looked at patterns of genetic diversity in populations impacted by humans (102, 103). A meta-analysis of population genetics studies found decreases in genetic diversity in animal and plant populations under pressure of habitat fragmentation and no consistent signal for populations affected by hunting or fishing, but found diversity increases in populations affected by pollution (103). Another meta-analysis, targeted only at mammals, found significant lower genetic diversity in mammalian populations that have experienced a reduction in population size or range or a population bottleneck (102). In a rare longitudinal study, Lage & Kornfield (104) looked at the genetic diversity in a population of the Atlantic salmon (*Salmo salar*) using samples from 1963 to 2001. They found that genetic diversity declined during this period, closely following population declines.

5. UNDERSTANDING THE DIRECT PRESSURES

We now examine five major categories of global biodiversity change pressures. For three of those, habitat change, pollution, and climate change, global models of their impacts are available, and we make comparisons between the terrestrial spatial pattern of the driver and the impacts on species extinction risk (**Figure 7**). Note, however, that the current biodiversity impacts of land-use change are much greater than the impacts of the other two drivers (**Figure 3**).

5.1. Habitat Change and Degradation

Habitat change and habitat degradation are currently the major drivers of global biodiversity change (**Figure 3**). In terrestrial systems, land-use change dynamics can be broadly classified into three categories: conversion of natural habitats to human-dominated habitats, intensification of human use of human-dominated habitats, and recovery of natural vegetation and forest in areas that have been previously cleared by humans. Not all species respond equally to habitat changes (105–107): When forest is converted to agriculture and pastures, some species may increase in abundance, whereas other species, particularly habitat specialists (108, 109), can decline or even go locally extinct.

Although the three types of land change dynamics occur in most world regions, the relative importance of each one has a strong latitudinal pattern (**Figure 7*b***) (2, 110): Most conversion of natural to human-dominated habitats is occurring in tropical forests (111); agricultural intensification started in the developed regions but is rapidly expanding to the rest of the world (not represented in **Figure 7*b***) (112); most recovery of natural and forest vegetation is occurring in temperate regions in Europe and North America (17) (**Figure 7*b***). A net forest loss of about 42,000 km^2 per year (111) in tropical regions is partially balanced by a net forest gain of 8,700 km^2 per year in Europe (110). However, part of the net forest gain is the result of new forest plantations, often with exotic species, which often have lower biodiversity than natural forests (113). Fire plays a major role in many regions in the conversion of forest to agriculture but also in maintaining open landscapes.

As expected, there is an agreement between the spatial distribution of areas of natural habitat being converted to agriculture and the distribution of species affected by habitat loss (**Figure 7*a*,*b***), including in Madagascar, some areas of sub-Saharan Africa, Brazil's Atlantic Forest, the Middle East, and Southeast Asia. Forest loss in Southeast Asia is not well captured in our land-use change map but has been reported in other studies (57). There are some regions where there is a high proportion of species affected by habitat loss where most land-use change already occurred in the past (much of Europe), and regions where species have been affected by habitat loss not captured in our analysis (e.g., the Sahara).

River systems have been deeply altered by impoundments and diversions to meet water, energy, and transportation needs of a growing human population (14). Today, there are more than 45,000 large dams (>15 m in height) worldwide (14). Dams have upstream impacts, where lotic systems are changed into lentic systems, and downstream impacts, where the timing, magnitude, and temperature of water flow is changed (45). Dams are also responsible for the fragmentation of river systems, as they hamper or even block the dispersal and migration of organisms (14). Furthermore, water resource development by impoundments and diversions has high spatial overlap with other pressures in freshwater ecosystems, such as pollution and catchment disturbance by cropland (114). Other important habitat changes in freshwater ecosystems include the loss of wetlands owing to drainage for conversion to agriculture or urbanization, overextraction of groundwater (45), and the excavation of river sand (115).

Marine habitats are also being affected by human activities, particularly by destructive fishing practices, such as trawling and dynamiting (116). Coastal habitats and wetlands have been affected mostly by urbanization, aquaculture development, and coastal engineering works (15, 77).

5.2. Overexploitation

Overexploitation is the major driver of biodiversity loss in the oceans (2, 19). Capture fisheries production increased for much of the twentieth century but has reached a plateau since the mid-1980s at around 70–80 million tons annually, despite continuing increases in global fishing effort levels (117, 118). The global landings would have likely declined except for the spatial expansion of the fishing effort toward deeper and further offshore waters. By the mid-1960s, most fully exploited or overexploited fisheries were located in coastal areas of the Northern Hemisphere. By the 1980s, fishing efforts were having an impact on regions much farther away from the coast, in the middle of the northern and southern Atlantic Oceans. One decade later, the spatial expansion of the fisheries had reached much of the world's oceans, with only some parts of the Indian Ocean, the Pacific Ocean, and the Antarctic ocean not having reached maximum historical catches (116).

In terrestrial systems, hunting is a major concern in tropical savannahs and forests (2). Large birds and mammals are targeted for their meat and charismatic species for their ornaments and alleged medicinal purposes (108, 111). Wild-meat harvest has been estimated at 67–164 thousand tons in the Brazilian Amazon and 1–3.4 million tons in Central Africa (119). The impacts are particularly acute in Southeast Asia and Central Africa (111). A connection has been established between the reduction of fish availability per capita and the increase in hunting pressure of wild meat in West Africa (120). Synergistic interactions between hunting and other drivers, such as land-use change and disease, can also occur and cause local extinctions (106).

5.3. Pollution

Eutrophication and other ecosystem changes caused by pollution are major drivers of biodiversity loss and alterations in both inland waters and coastal systems (121). River nitrogen loads from point sources, such as domestic and industrial sewage, and nonpoint sources, such as agriculture and atmospheric deposition, increased in most world regions from 1970 to 1995 but are starting to decline or are projected to decline until 2030 in Europe and northern Asia (Russia) (122). Lakes are particularly vulnerable to regime shifts caused by eutrophication, which may be difficult to reverse (47, 123). Eutrophication can lead to increased biomass of phytoplankton and macrophyte vegetation, blooms of toxic cyanobacteria and other algae, higher incidence of fish kills, and, in the case of coral reefs, declines in coral reef health and loss of coral reef communities (121).

Atmospheric nitrogen deposition from intensive agriculture and fossil-fuel combustion can also affect terrestrial ecosystems, particularly temperate grasslands (2). The increase in availability of nitrogen changes the competition dynamics in plant (124) and lichen communities (125), favoring the increase of nitrophilous species and the decline of nitrogen-sensitive species. One study found a linear relationship between the rate of nitrogen deposition and species richness declines in temperate grasslands and estimated that, for the levels of nitrogen deposition observed in much of central Europe (17 kg/ha/year), a 23% reduction of species diversity can be expected (124). Unfortunately, some high species diversity regions (e.g., Southeast Asia and Brazil's Atlantic Forest) are also receiving similar levels of nitrogen deposition (**Figure 7*d***), but more research is needed to identify its impacts (126). A visual inspection of the spatial overlap between the global patterns of nitrogen deposition and the distribution of vertebrates affected by pollution shows reasonable agreement in Europe, but inspection also shows disagreement in other parts of the world, such as Central Africa (**Figure 7*c*,*d***). Note, however, that there are other sources of pollution included in the assessment of species extinction risk (**Figure 7*c***) and not directly related to atmospheric nitrogen deposition (**Figure 7*d***).

5.4. Introduction of Exotic Species and Invasions

One of the major trends in global biodiversity change is the increased homogenization of plant and animal diversity owing to biotic exchange. In some cases, exotic species are able to spread beyond the places where they were introduced, spreading in the landscape and outcompeting native species (127). Islands have been particularly affected by invasive species (128): Animal invasions have led to species extinctions, whereas plant invasions can decrease the abundance of native species and become dominant in plant communities. Plant invasions may also affect the nutrient cycles, alter the fire regimes, and impact other ecosystem services (129, 130). A particularly serious type of invasions is epidemic disease. One example is chytridiomycosis, which has been decimating amphibians in many regions of the world and is a leading cause of the global amphibian decline (131). Invasive species have also had important impacts on freshwater ecosystems, where their incidence is correlated with human economic activity (132), and in marine and estuarine ecosystems due to ballast water or hull fouling transported by ships (133).

Still, many invasive species have had more moderate impacts on ecosystems (134), and recently, some ecologists have called for a more embracing attitude toward exotic species, arguing that alien species should not be a priori considered negative in an ecosystem but should be assessed objectively for their impacts (135, 136). Others have argued for active translocation or assisted migration of species endangered by climate change (137), an approach that seems fraught with peril on the basis of our historical experience of human introductions of exotic species, often with the best intentions.

5.5. Climate Change

Global mean surface temperature increased 0.74°C from 1906 to 2005 and is expected to increase between 1.8°C and 4°C during the twenty-first century, depending on the socioeconomic scenario (138). Warming is spatially very heterogeneous as it is largest in terrestrial systems and at high northern latitudes, with recent warming greater than 1.5°C in some areas, and least pronounced in the tropics, where many regions have warmed around 0.5°C (**Figure 7*f***). The impacts of climate change are already contributing to increased extinction risk of species at high northern latitudes (**Figure 7*e***). Further climate change impacts in these regions have been projected for birds (139) and for plants (46) during this century. Surprisingly, in the Cape region (South Africa) and in southeastern Australia, a high incidence of species negatively affected by climate change has been reported (**Figure 7*e***), although these areas are not suffering large warming (**Figure 7*f***). One explanation may be that those regions have many species particularly vulnerable to climate change. Species with high vulnerability are species that have narrow climate niches, cannot shift their ranges, or are unable to change their phenology, evolve their physiology, or behaviorally adapt to the new conditions (93, 140). For instance, the limited ability of mountaintop species to shift in elevation has been identified as a major climate vulnerability (92).

For amphibians, important future climate impacts have been projected in the northern Andes, parts of the Amazon, Central America, southern and southeastern Europe, sub-Saharan tropical Africa, and Southeast Asia (140, 141). Surprisingly, this disagrees somewhat from the recent spatial patterns of increased extinction risk owing to climate change (**Figure 7*e***).

In corals, most threatened and climate change–susceptible species occur in Southeast

Asia (140). Climate change is also causing sea-level rise and threatening coastal habitats, particularly in synergy with land-use change, which may not allow coastal habitats to migrate inland (47). Marine ecosystems are also affected by ocean acidification caused by climate change, particularly corals (79) and other marine organisms that build calcium carbonate skeletons (142).

6. EXPLORING THE UNDERLYING CAUSES WITH SCENARIO MODELS

Upstream from the direct pressures on biodiversity, there are indirect drivers of biodiversity change. Major indirect drivers for biodiversity include population growth, energy use and energy production, diet, and food demand. Naturally, these drivers interact between them and with other drivers, such as technology development, socioeconomic changes, and cultural transformations (143). One way of exploring the relationship between the indirect drivers and global biodiversity change is through scenario modeling. Biodiversity scenarios have recently been reviewed elsewhere (3). They can be developed in three steps: (*a*) plausible trajectories of key indirect drivers are generated; (*b*) the trajectories are fed into models that project changes in direct pressures; and (*c*) projected pressures are used as inputs of biodiversity models. Many scenarios explore different futures and how they depend on policy decisions, but scenario models can also be used for hindcasting, i.e., to reconstruct the past.

The human population increased from 2.5 billion people in 1950 to 7 billion in 2011 and can reach between 8.1 billion and 10.6 billion people in 2050, depending on the scenario (144). The increase in human population growth is being accompanied by an increase in the demand for food (with food production growing faster than human population) and an increase in energy consumption (2). How much of the increase in food production needed over the next few decades will come from intensification or from farmland expansion to natural habitats will depend on technological developments, policy choices, and societal behavior. Similarly, how a growing energy demand will be satisfied by additional fossil-fuel consumption or by shifting energy production toward other sources has also been explored in scenarios.

Most scenarios project a decrease in forest area by 2050 of up to 20% and in an extreme case, of more than 60% (3). Still, some scenarios that account for policies recognizing the role of forests in CO_2 sequestration and avoiding the impacts of land-use changes, including conversion of forests to biofuels, project net increases in forest area (3). Species extinction rates will continue to be higher than in the fossil record. For the same modeling approach, scenarios with lower levels of population growth and climate change result in lower estimates of biodiversity loss.

7. A BIT OF GOOD NEWS FOR A CHANGE

Despite the gloomy biodiversity picture depicted in the previous sections, there is also some good news about global biodiversity change due to the reversion of the effect of a driver (e.g., forest recovery) or the successes of conservation initiatives on the status of species (**Table 1**).

Measures such as habitat conservation, reintroduction programs, and legislation have proven to be efficient in improving the status of several species (145). One way to assess conservation successes is to identify prevented extinctions. Between 1994 and 2004, 16 bird species would have gone extinct if actions had not been undertaken to protect them (146). One example is the population of the Norfolk Island green parrot (*Cyanoramphus cookii*), very likely to go extinct in 1994, with only four breeding females, which has now close to 300 individuals thanks to habitat protection and control of predator and competitor species. In Europe, a comparison of bird population trends between Birds Directive Annex I (higher protection level) and non-Annex I species

Table 1 Examples of successful outcomes of global, regional or national conservation initiatives (expanded from References 8 and 54)

Successes[a] (references)	Detail/examples
Improvement in the Red List classification of species (8, 54, 145)	Mammals: 24–25 species out of 195 between 1996 and 2008 (1 species for every 7 with decreasing status) Birds: 33–44 species between 1988 and 2008 Amphibians: 4–5 species between 1980 and 2004 The improvement in the conservation status of these species is explained by habitat protection, reintroduction programs, legislation, control of competitors, or a combination of those measures.
Impact of the Bird Directive in Europe: Annex I listing (147)	Birds: significantly higher population trends for the 1990–2000 period when comparing Annex I and non-Annex I species.
Prevention of species extinction (146)	Birds: extinction was avoided for 16 species classified as critically endangered by the IUCN. The mean population size for these species was augmented from 34 individuals in 1994 to 147 in 2004. Conservation measures included habitat-based protection, invasives control, captive breeding, and (re)introductions.
Natural recolonization and recovery from local extinctions (153)	Mammals: increasing population size and distribution for carnivore species between 1970 and 2005 in Europe, following land abandonment and reduced human pressure: Gray wolve (*Canis lupus*) from 8,000 to 18,500 individuals; Brown bear (*Ursus arctos*) from 10,000 to 14,000 individuals; and Eurasian lynx (*Lynx lynx*) from 4,000 to 8,000 individuals.
Increased Water Quality Index (8)	This index of the physical and chemical quality of freshwater increased 7.4% in Asia between 1980 and 2005.
Restored fishery stocks (19)	In parts of the coasts of Australia, Canada, Iceland, New Zealand, and the United States, recovery of fishery stocks was made possible by the implementation of management programs designed to lower fishing pressure, to prevent overfishing, and to restore marine ecosystems.
Decreased pressure on forests (8)	In 2008, the annual area deforested in the Amazonian forest of Brazil represented less than half of the area cleared in 2004 (1.3 million ha versus 2.8 million ha). Nonetheless, it is not clear whether this decrease is due to legislation or to less demand for natural resources.
Conservation status and population trends in the EU25 (148)	Birds: 12 species (out of 448) no longer have an unfavorable conservation status (228 in 1990 versus 216 in 2000). Increasing population trends were also observed for species in marine, coastal, inland wetland, and Mediterranean forest habitats.

[a]Abbreviations: EU25, European Union member states as of 2004; IUCN, International Union for Conservation of Nature.

(lower protection level) also shows significant differences, highlighting the effectiveness of the European policies (147). The conservation of threatened habitats, such as inland wetlands and Mediterranean forests, also allowed for an increase in some bird populations trends (148). At a global scale, Hoffman et al. (54) identified 68 species, including 40 birds, 4 amphibians, and 24 mammals, that showed an improvement in their conservation status, leading to a revision of their IUCN category. In the marine biomes, the restoration of fishing stocks can deliver important benefits (19).

Conservation successes are also associated with the implementation of protected areas. Protected areas considerably increased during the past century and now cover 12% of the terrestrial surface (8). However, designations of protected areas do not always lead to the implementation of on the ground effective measures to protect habitats and species (149). Complementary tools to combat declines of biodiversity outside nature reserves are agri-environment schemes, which are policy tools with ample scope to reverse the negative trends of once-common, widespread species (150), and direct payments to conserve biodiversity (151).

In some regions of the world, a habitat conservation strategy that is emerging as a significant opportunity is the rewilding of abandoned areas (17). Natural revegetation in large areas has been observed in the past and is predicted to occur in the next decades (**Figure 7*b***), particularly on remote and marginally productive areas (e.g., mountains) in the Northern Hemisphere where agriculture and forestry activities are being abandoned (46, 152). The subsequent reappropriation of the land by wildlife can be beneficial for various species that take advantage of the reduced human pressure (17): Several European carnivores have been coming back to countries where they were previously extinct (153). Still, natural regeneration presents certain challenges that depend on the level of resilience of the land (154, 155) and potential conflicts with human populations (17).

Finally, aside from avoided extinctions and increasing population trends, conservation successes can be measured in changes in societies' behavior regarding sustainable resource uses. The increasing public support for biodiversity conservation in the past few decades (156), the commitment of the Convention on Biological Diversity parties to new goals for 2020 (157), and the recent establishment of the Intergovernmental Platform on Biodiversity and Ecosystem Services (**http://www.ipbes.net**) give hope of further progress in the years to come.

8. MAJOR GAPS IN OUR UNDERSTANDING OF GLOBAL BIODIVERSITY CHANGE

In this article, we reviewed the current scientific knowledge about the state of global biodiversity change. Overall, the patterns that emerge allow us to state with confidence that biodiversity is being rapidly altered on land, in rivers, and in oceans, and is being lost locally in many regions and also globally. Some conservation actions have been successful at mitigating or, in a few cases, reversing biodiversity loss. However, many unknowns remain, and we still do not know the exact dimensions of the biodiversity crisis.

Some of the biodiversity indicators that were described in Section 4 and that were used to assess the 2010 target of the Convention on Biological Diversity are far from being completely developed (149). Very little is known about trends in genetic diversity, particularly in wild species. The taxonomic coverage of the indicators and assessments is very limited: The extinction risk of the vast majority of biodiversity is not known (**Figure 4*a***), and most of the population indicators are derived from vertebrate populations (**Figure 4*b***). This is not to say that the same conservation and research emphasis shall be placed on all biodiversity. People place high existence values on vertebrates (158), and many important ecosystem services are associated with vertebrates (48, 62). It is just an acknowledgment of the large gap in our taxonomic knowledge of global biodiversity change. More worryingly, even the available information for vertebrates is spatially very heterogeneous (72) and is least available in regions that are currently under pressure (**Figure 8**). The Group on Earth Observations Biodiversity Observation Network (GEO BON) aims at filling these gaps by integrating biodiversity monitoring programs across the globe and promoting biodiversity monitoring in gap regions (159).

Major gaps and uncertainties remain in modeling global biodiversity change. In terrestrial systems, most research has been dedicated to model climate change impacts, although some work has also been done on modeling the impacts of land-use change and, to a lesser extent, pollution. Models are lacking for the global spatial distribution of exploitation pressure and invasive species and their impacts in terrestrial systems. But even for the pressures that have received most attention, large uncertainties remain: Projected extinction rates for this century range from less than 20 E/MSY to more than 14,000 E/MSY (3). Both the lack of harmonization of modeling approaches and the lack of knowledge of how species respond to global change contribute to this uncertainty.

9. IS ALL BIODIVERSITY CHANGE EQUALLY BAD?

Our world is changing, and biodiversity is no exception. Yet, not all biodiversity change is inherently bad, and we should avoid a static view of conservation biology (despite its name). The maintenance of the landscapes or the biological communities we know should not be the a priori management target. We need to assess biodiversity change with objective criteria. The ecosystem services framework (2, 70), with the appropriate inclusion of species existence values (158), is an excellent tool to assess the management priorities for biodiversity change. It allows us to identify not only the benefits and costs of biodiversity alterations for human well-being but also to prioritize the biodiversity losses that are more important to address.

Biodiversity alterations and losses have to be assessed for their contribution to ecosystem processes, such as nutrient cycling and soil formation, and to ecosystem services, such as climate regulation, water quality regulation, water provisioning, timber provisioning, disease and pest regulation, and cultural services. An appropriate inclusion of existence values is essential; people place large values on the conservation of particular species or taxonomic groups. Therefore, not all species extinctions can be treated equally from a utilitarian point of view. The extinction in the wild of the viruses variola major and variola minor, the causes of the deadly smallpox, was arguably a good thing. But in many more cases, the loss of biodiversity is impoverishing us and making our planet more unequal for its human inhabitants: It is often the poor that suffer the first negative impacts of biodiversity change (2).

Some biodiversity alterations, such as the conversion of the Amazon forest to agricultural areas, may lead to tipping points in ecosystems that are hard to reverse (47), but the majority of biodiversity alterations are reversible through management. In contrast, biodiversity loss is usually irreversible: Extinction is forever, at least with the current biotechnology level. Scientists can inform society about how biodiversity is changing and what the likely consequences of those changes are for ecosystems and for human well-being, but it is up to society to decide what should be done about these issues.

SUMMARY POINTS

1. Biodiversity change is composed of biodiversity loss, such as species extinctions, and biodiversity alterations, such as species range shifts.
2. Biodiversity is changing at unprecedented rates in human history: Species are becoming extinct or closer to extinction; mean species abundances of several taxa are decreasing; species are shifting their ranges in response to climate change; and domestic and wild genetic diversity are being lost.
3. The major direct drivers of biodiversity change are habitat change and overexploitation. Pollution, exotic species, and disease are also important drivers. Climate change is an emerging driver of biodiversity change.
4. Human population growth and human resource use are the underlying indirect drivers of biodiversity change.

5. There have been some important successes in biodiversity conservation—mainly through species management, protected areas, and increased societal awareness. Farmland abandonment is an opportunity for biodiversity restoration.
6. Not all biodiversity change is bad. Biodiversity change should be assessed in relation to its consequences for ecosystem services and species existence values.

FUTURE ISSUES

1. There are major gaps in our knowledge of biodiversity change, and there is the need to improve our biodiversity monitoring programs worldwide.
2. There are also important uncertainties and gaps in our models of global biodiversity change.

DISCLOSURE STATEMENT

The authors are not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

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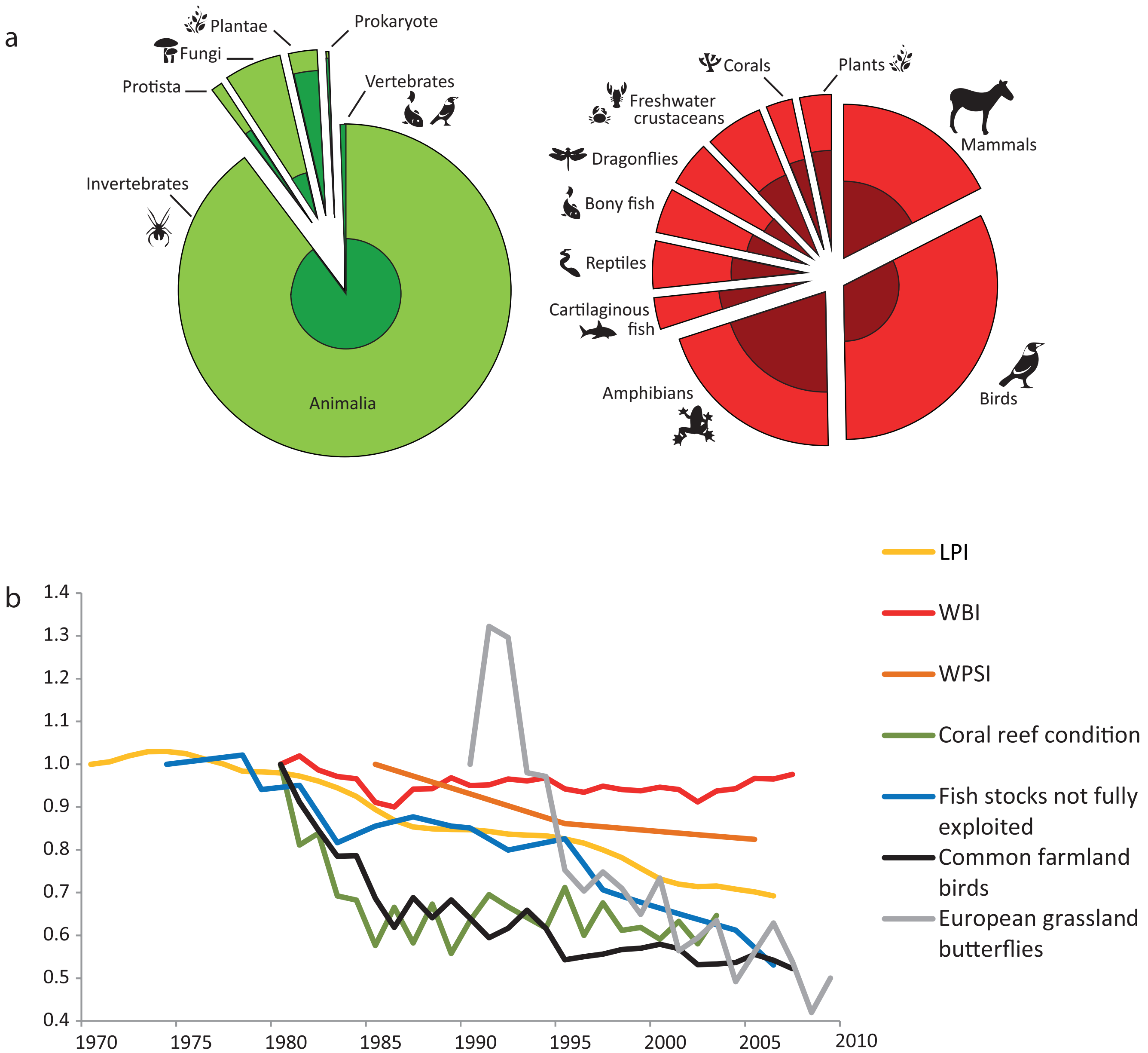


Figure 4

(*a*) (*left*) Estimated proportion of species in each of the main domains (52). For each taxonomic group, the dark green identifies the proportion of species already described relative to the estimated number of species. (*right*) Proportion of assessed species for each taxa that has been representatively evaluated by the IUCN (plants include cycads, conifers, and sea grasses; freshwater crustaceans include crayfish and crabs). The dark red identifies the proportion of species threatened in each group (54). (*b*) Evolution of some of the main biodiversity indicators between 1970 and 2010 (8, 162, 163). All indicators are dimensionless as they are scaled relative to their values in the first year for which information is available. (*c*) Observed northward shifts in species or communities of species (km/year): ① meta-analysis of shifts of the northern range limit for 99 species of butterflies, birds, and alpine herbs in Europe (mean ± standard error) (85); ② northward shift in the composition of bird and butterfly communities in Europe (mean ± standard error) (90); ③ mean shift of the northern range limit for 16 taxa in the United Kingdom, based on heavily recorded atlas cells (lower bound), well-recorded cells (middle line), all recorded cells (higher bound) (89); ④ mean shift of the northern range limit for 28 species of bottom-dwelling fishes in the North Sea, for all species (middle line), for warm specialists (upper bound), and for cold specialists (lower bound) (83). (*d*) Risk status for breeds of mammalian (5,600 breeds) and avian (2,000 breeds) domesticated species (96). The "at risk" category includes critical, critical-maintained, endangered, and endangered-maintained species. Abbreviations: LPI, Living Planet Index; WBI, Wild Bird Index (of habitat specialists); WPSI, Waterbird Population Status Index.

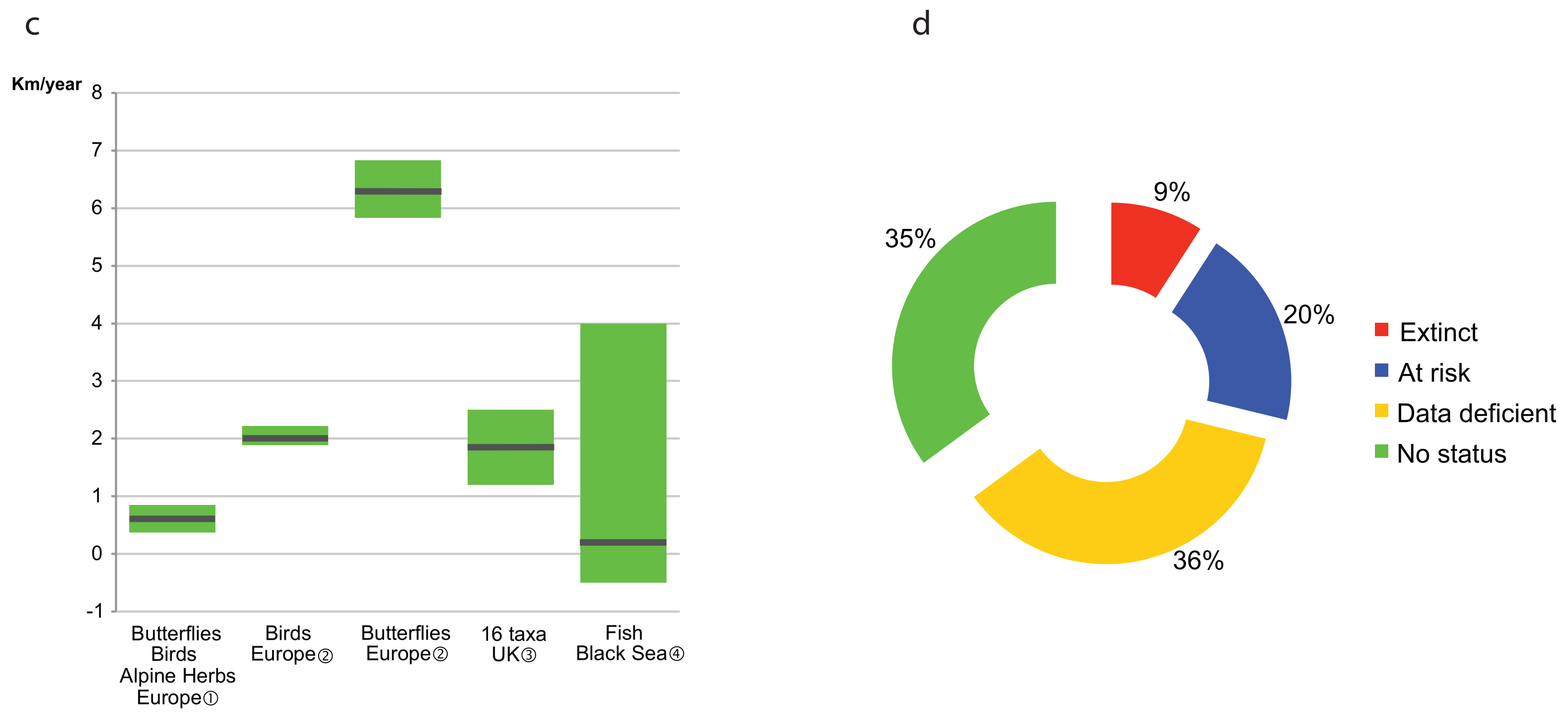
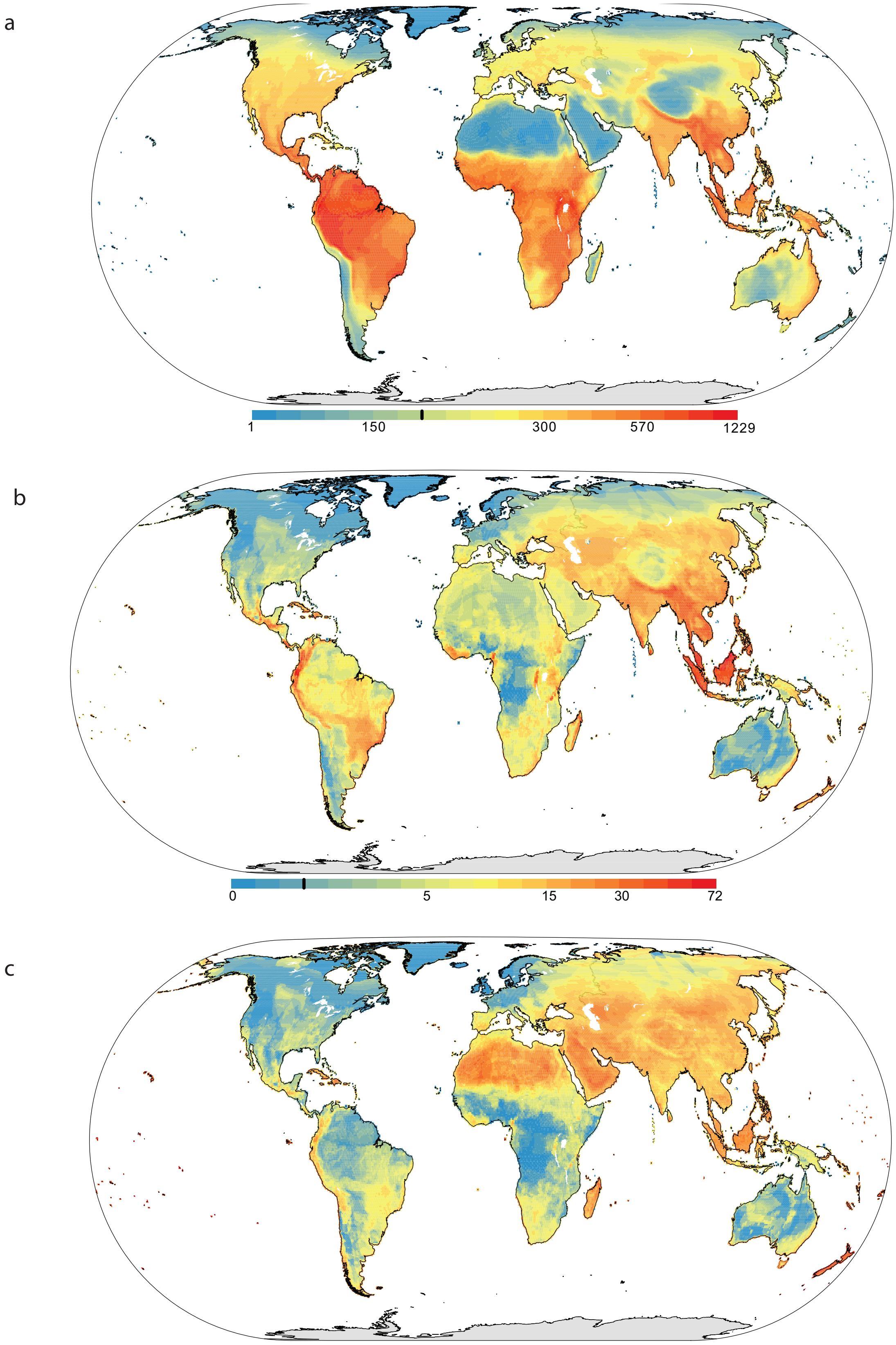


Figure 4

(*Continued*)


a
1
150
300
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0.15
0.35

Figure 5

Global distributions of terrestrial vertebrates and threatened species, based on species ranges for birds ($n = 10{,}606$) (164), mammals ($n = 5{,}348$), and amphibians ($n = 6{,}248$) (161). Color scales are based on geometric intervals (interval size increases at a constant ratio to the left and to the right of the black bar in the scale). Density calculations are based on grid cells of $0.48^\circ \times 0.48^\circ$. (*a*) Species richness. (*b*) Number of threatened species (critically endangered, endangered, or vulnerable). (*c*) Proportion of threatened species (number of threatened species divided by number of species in each cell).

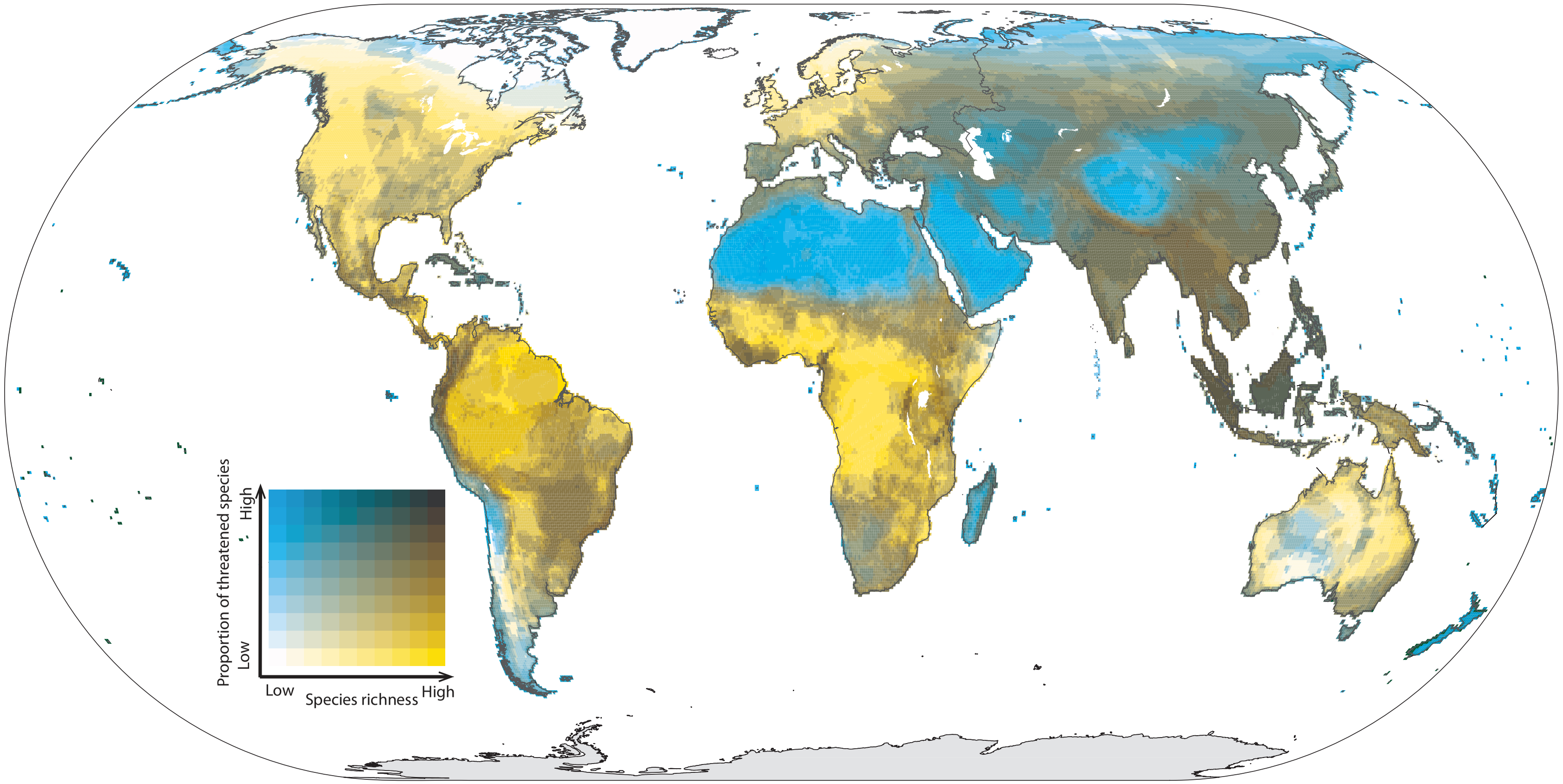
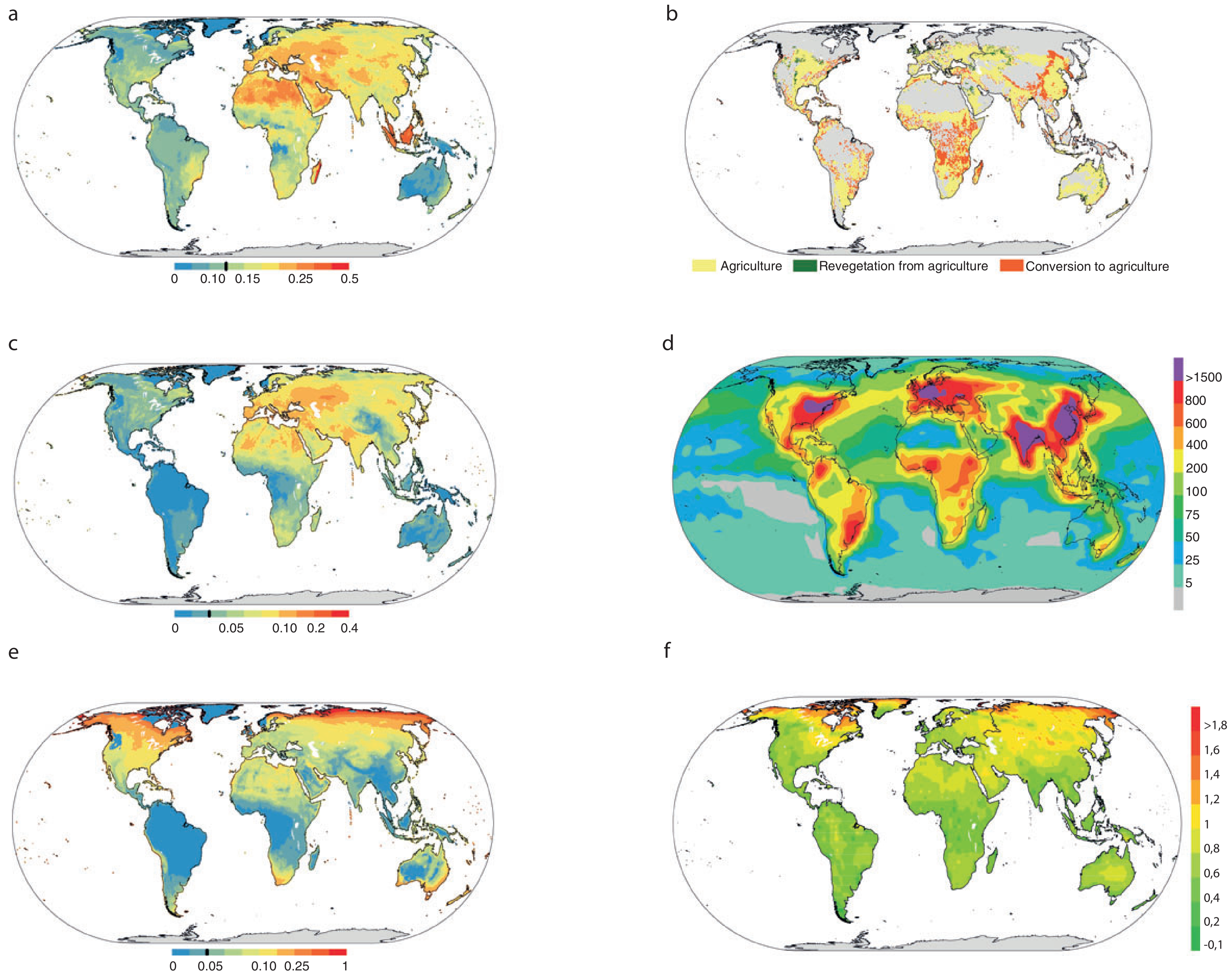


Figure 6

Global distribution of the overlap between terrestrial vertebrate density and incidence (proportion) of threatened species (161, 164). Bright yellow areas have high species richness but low threat; bright blue areas have low species richness but high threat; dark areas have high species richness and high incidence of threatened species; light yellow and light blue areas have low diversity and low threat.


a
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b
Agriculture
Revegetation from agriculture
Conversion to agriculture
c
0 0.05 0.10 0.2 0.4
d
>1500
800
600
400
200
100
75
50
25
5
e
0 0.05 0.10 0.25 1
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>1,8
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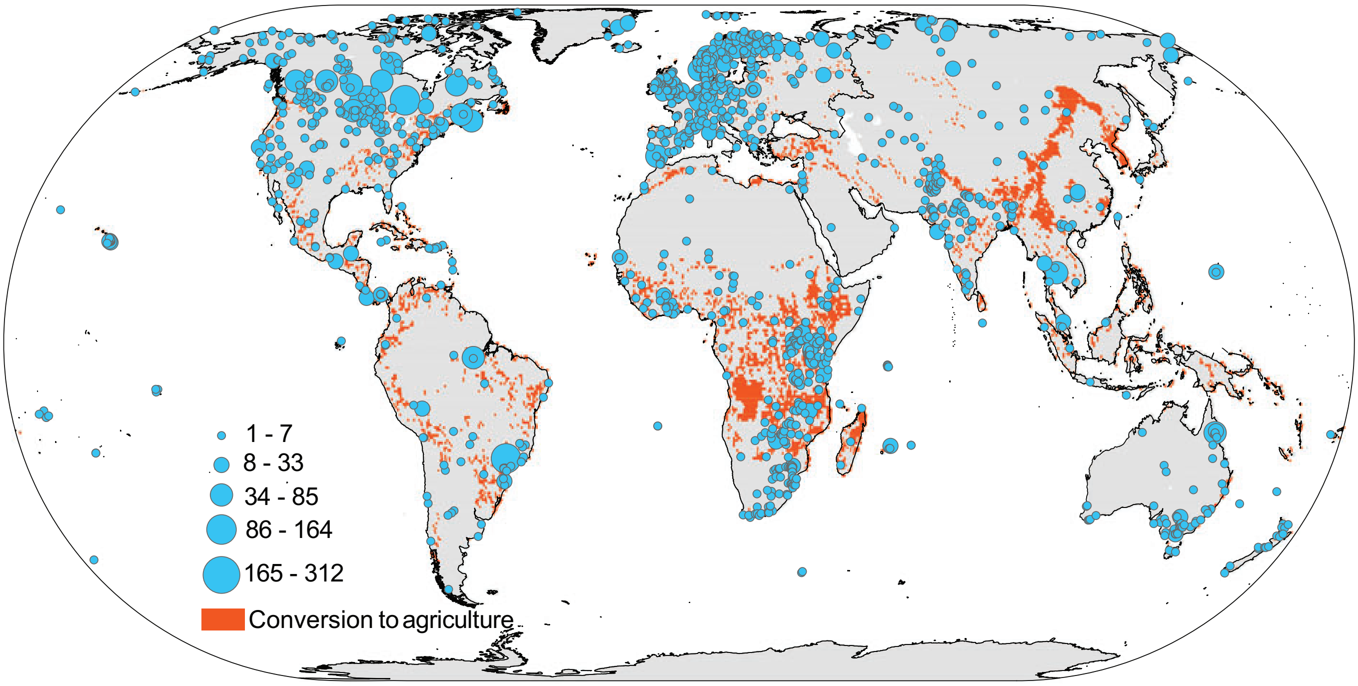

Figure 8

Overlay between the predicted loss of natural habitat and the distribution of the populations monitored by the Living Planet Index (LPI). The circles illustrate the geographical origin of the data used to calculate the annual LPI (20). The size of these points varies according to the number of populations being monitored. The map of land-use change represents the areas of conversion from natural habitat to agriculture, based on the projections of the Order from Strength scenario between 1970 and 2020 (165).

Figure 7

Global distribution of impacts of drivers on terrestrial vertebrates (*panels a,c,e*) and the intensity levels of those drivers (*panels b,d,f*). The impacts include all species listed in the International Union for Conservation of Nature (IUCN) Red List as negatively affected by those drivers, including threatened and nonthreatened species (161). (*a*) Proportion of species suffering from habitat loss (residential and commercial development, agriculture and aquaculture, energy production and mining, transportation and service corridors, and natural system modifications). (*b*) Land-use change between 1970 and 2020: revegetation from agriculture, conversion from natural habitat to agriculture or steady agricultural use. This panel is based on the projections of the Order from Strength scenario (165). (*c*) Proportion of terrestrial vertebrates suffering from pollution. (*d*) Nitrogen deposition (in milligrams of nitrogen/m^2/year) in 1993 (166). (*e*) Proportion of species suffering from climate change and severe weather. (*f*) Annual mean surface temperature change between the average of 1965–1975 observations and the average of 2015–2025 model projections for the Intergovernmental Panel on Climate Change B1 scenario (167).

B. Appendix: Beilin et al. 2014

Beilin, R., Lindborg, R., Stenseke, M., Pereira, H.M., Llausàs, A., Slätmo, E., Cerqueira, Y., **Navarro, L. M.**, Rodrigues, P., Reichelt, N., et al. (2014). *Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from Scandinavia, Iberia and Oceania.* Land Use Policy 36: 60–72.

In this publication, Laetitia M. Navarro participated in the description of the Portuguese case studies and in the analysis of the Portuguese data on land-use change and biodiversity. LMN also took part in the definition of the pressures, frictions and attractors of land abandonment. The ideas for the manuscript were developed during a workshop held in Poowong, Australia in october 2010, funded by the Swedish Research Council for Environment, Agricultural Sciences and Spatial planning (FORMAS/LUPA).

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Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from Scandinavia, Iberia and Oceania

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ABSTRACT

Agricultural land abandonment (ALA) is widespread in many countries of the global north. It impacts rural communities, traditional landscapes, biodiversity and ecosystem services. It is an opportunity for ecosystem restoration or new landscape functions. We explored ALA in study areas in Australia, Portugal and Sweden. In each, we assessed plant species diversity, historical trajectories of land cover change; and the socioeconomic past, present and future in interviews with farmers. The ALA data was integrated and analysed by identifying the drivers of change. The relative importance of each driver and its scale of action was estimated, both in the past (1950–2010) and in the future (2010–2030). ALA has transformed rural landscapes in the study areas of Portugal and Sweden. It is at a much earlier stage with potential to increase in the Australian case. We identified a set of driving forces, classified into pressures, frictions and attractors that clarify why ALA, noting its temporal and spatial scale, occurs differently in each study area. The effect of the drivers is related to social and historical contexts. Pressures and attractors encouraging agricultural abandonment are strongest in Portugal and Sweden. Generally more (institutionalized) frictions are in place in these European sites, intended to prevent further change, based on the benefits assumed for biodiversity and aesthetics. In Australia, the stimulation of driving forces to promote a well-managed abandonment of some cleared areas could be highly beneficial for biodiversity, minimally disruptive for current dairy farming operations and would bring opportunities for alternative types of rural development.

Introduction

Land use change has implications for sustainable development at a global scale (Pereira et al., 2010), particularly with regard to feeding a growing and wealthier human population (Foley et al., 2005). During the last 50 years there have been two main land use trajectories affecting biodiversity and nature values in agricultural landscapes (Tilman et al., 2001). More economically productive areas have been intensified, incorporated into larger assemblages particularly within developed countries (Stoate et al., 2009), whereas remote, economically unproductive farm areas are increasingly abandoned, reforested, or included in rewilding for nature values with the creation of nature reserves or parks (Pinto-Correia, 1993; MacDonald et al., 2000; Navarro and Pereira, 2012). Although the connections between social and ecological values are recognized as important (Folke, 2006; Carpenter et al., 2009), few studies integrate both social and ecological data from case studies

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to examine the links between social drivers and their effect on land use and biodiversity. Yet, the trend in land abandonment evinces concern and/or opportunity globally but for markedly different reasons in various landscapes (Moreira et al., 2001a; Gellrich et al., 2007; Cramer et al., 2008; Cetinkaya, 2009).

In this study we compare case studies in Australia, Portugal and Sweden. These countries have had different social-economic development during the last century, and all three experience degrees of agricultural land abandonment (ALA). In each country we examine the drivers of change affecting ALA and discuss effects on land use and on the social and ecological systems. In doing so, we will integrate what Lambin et al. (2001) refer to as local-level case studies with regional 'generalities' to link to international themes of global change. The goals of our study are to examine: (i) which are the main drivers influencing management practices associated with agricultural land use change and abandonment in the three locations; (ii) what are the temporal and spatial scales and institutional levels framing land use change; and (iii) how have these drivers, through different land use changes, affected and will affect biological and cultural outcomes. Ultimately, this framework is intended to enable and better support policy decisions associated with agricultural land abandonment.

Agricultural land abandonment

The degrees of land use change have both spatial and temporal historicity, associated with the decline in agricultural productivity within regions, across economies and in association with technological and demographic changes (Beilin et al., 2011). ALA is commonly defined qualitatively as land condition; and quantitatively as years without agricultural use (Moravec and Zemeckis, 2007, p. 5). It may be evident at a variety of scales and with different degrees of intensity (Pinto-Correia, 1993; Burel and Baudry, 1995). At the extreme, abandonment is the complete withdrawal of agricultural management from the landscape and a transition to various non-agricultural lands uses (Keenleyside and Tucker, 2010). In other instances there is a reduction in management intensity, e.g. conversion of croplands to pasture areas (Bielsa et al., 2005). These situations may occur at different scales (MacDonald et al., 2000); and depending on spatial and temporal scales, the outcomes of abandonment may not be permanent (Wood, 1993).

ALA is driven by a combination of different factors, ranging from physical constraints of the landscape to more economic and social drivers (Strijker, 2005; Kizos and Koulouri, 2006; Koulouri and Giourga, 2007). A combination of these drivers is generally more pronounced in remote and isolated regions, such as mountain zones or marginal areas for agriculture where productivity is low, e.g. areas with shallow soil, salty soils, or poorly drained, while mechanization is often restricted by the terrain (MacDonald et al., 2000; Gellrich and Zimmermann, 2007; Cramer et al., 2008).

The diminishing of agriculture in an area often co-varies with socio-economic changes and has ecological consequences, which can differ with local land use history, climate and landscape composition (Rey Benayas et al., 2007). Related socio-economic changes in remote areas are characteristically demographic, for example, declining and ageing populations. How biodiversity is affected by different land use changes is a contested topic in conservation research (Balmford et al., 2005; Matson and Vitousek, 2006; Dorrough et al., 2007; Vandermeer and Perfecto, 2007) and that is true in relation to land abandonment (Pereira et al., 2005; Navarro and Pereira, 2012). Agricultural practises have been responsible for the destruction and fragmentation of many native habitats with consequent negative impacts on biodiversity (Poschlod et al., 2005); therefore it could be expected that retiring agricultural land from production can be an opportunity to improve the condition for many species that were severely affected by native habitat fragmentation in the past (Reidsma et al., 2006; Foster et al., 2009; Aide et al., 2012). However, decline in traditional low-intensive agriculture often has negative effects on biodiversity at the local scale, as the moderate utilization of grasslands and forests in historical times were associated with high species richness in the rural landscape of Europe (Kull and Zobel, 1991; Eriksson et al., 2002; Pykälä, 2004), leading to the concept of high nature value farmland (Halada et al., 2011).

Drivers of change

'Driving forces' are defined as 'the forces that cause observed landscape changes, i.e., they are influential processes in the evolutionary trajectory of the landscape' (Bürgi et al., 2004, p. 858). The potential of the concept lies in the possibility of using driving forces as a theoretical frame to understand and analyse changes and also, by comparing case studies, to identify common land use trends (Eiter and Potthoff, 2007; Schneeberger et al., 2007).

In a literature review of the concept and application of driving forces of land use change, Slätmo (2011) characterizes them into four concepts: pressures, frictions, attractors and triggers. *Pressures* are factors that are pushing for change, with long term implications that put a stress on the current land use. Pressures can be divided into different types deriving from overarching categories such as political, economic, cultural and technical. *Frictions* are factors that serve to resist change: preventing, slowing down or changing the direction of land use change. Frictions can be divided into different types deriving from the same overarching categories as 'pressures'. *Attractors* are associated with site conditions that attract change due to their privileged characteristics and/or location. *Triggers* are factors that spur land use change in a direct, time specific ways (e.g. the opening of a new road). Therefore, even if their impact is significant, they are difficult to relate to long term landscape transformations due to their short-term nature. If a trigger persists in time, then it can be easily classified as either a friction or a pressure (e.g. increased accessibility because of the new road).

Study areas

Individual sites in each country (Australia, Portugal and Sweden) incorporated landscapes exhibiting different levels of ALA, but sharing some overall similarities. These were the impacts of global markets, challenging biophysical conditions for farming and at least one hundred years of consecutive agricultural land use.

Poowong, Australia

The Australian case study is located in the south-eastern State of Victoria within a commutable 100 km of the capital city, Melbourne (Fig. 1). Encompassing approximately 18,000 ha of farmland predominantly used for grazing dairy cows and beef cattle, the research area is situated north-east of a small rural township, Poowong, spanning two local government administrations. The terrain is hilly and steep, but the elevation range is limited to between 90 and 300 m above sea level. The area is best described as a maritime temperate climate, with mean maximum temperatures around 24.5 °C in the summer and just above 13 °C in winter, with very rare episodes of minimum temperatures below 0 °C. The area receives a typically high annual rainfall by Australian standards, between 800 mm and 1400 mm. The combination of rugged topography and duplex soil types formed from sedimentary and basalt bedrock creates a landscape vulnerable to erosion and landslips through waterlogging and disturbance of vegetation cover for agriculture and forestry (Jenkin, 1970; Beilin, 2007). The study area forms part of the western section of the Strzelecki Ranges in the Strzelecki West Biodiversity Landscape zone. An estimated 600 people live

Fig. 1. The three countries, Australia, Portugal and Sweden, and the location of the respective study areas.

on independent holdings in the census area known as Poowong State Suburb (ABS, 2006).

Castro Laboreiro, Portugal

The Portuguese case study is located in the parish of Castro Laboreiro in the Peneda Mountain Range, in the north of the country (Fig. 1). The parish covers approximately 9440 hectares with elevation ranging from 300 m to 1340 m. Located at the transition between the Mediterranean and Atlantic biogeographic zones, the region has a temperate Mediterranean climate, characterized by temperatures varying between an average daily minimum of 0–3 °C in the winter months and an average daily maximum of 22.5 °C in the summer months. Precipitation exceeds 2000 mm per year, with relatively dry summers. The geology of the area reveals a strong presence of granite and quartzite, with thin and non-existent soils in the steepest slopes, whereas pluvial action formed deep soils in the valleys, nowadays intensively altered in their characteristics due to long human activity (ICN, 1995). The parish is in the Peneda-Gerês National Park and is part of the Natura 2000, the European Union protected area network. Over the last 50 years the population in mountain regions of Portugal has declined significantly and steadily (Pereira et al., 2005; Aguiar et al., 2009), and today Castro Laboreiro is inhabited by about 540 residents (INE, 2011). The rigorous winter and the orography of the parish led to the seasonal migration from summer villages (*brandas*) in the plateau to winter villages (*inverneiras*) in the valley. The parish consists of 15 brandas, 18 inverneiras and 8 fixed villages. The traditional migration of the local residents has steadily decreased since the early 1980s and now occurs only in a couple of villages. Farmers receive subsidies to maintain traditional farming and pastoral activities.

Hållnäs, Sweden

The Swedish study area (Hållnäs) is a peninsula in Southern Bothnian, situated in south-central Sweden in Uppland County (Fig. 1). The size of the parish is 25,000 ha. Hållnäs is located in an area affected by isostatic land-uplift (0.6 m/100 years), implying significant environmental changes in a relatively short time span. Currently the study area is within a range of 10–25 m above the sea level. In spite of the high northern latitude of Sweden, the climate can be classified as warm summer, with July being the warmest month (average maximum temperature of 21 °C), and January the coldest (with an average minimum of −8 °C), with freezing spells over consecutive days. Rainfall is higher during the summer months of the year (up to 60 mm/day), while less abundant in winter (up to 25 mm/day), averaging around 530 mm annually. Soils are nutrient poor, and cover a subsoil rich in calcium carbonate and bases. Natural values on the peninsula are to a large extent related to the uplift process. There are fourteen Natura 2000 areas and seven nature reserves encompassing shores, wetlands and coniferous forests on lime-rich soils (Naturvårdsverket, 2012). A large amount of the semi-natural pastures are acknowledged for their high biodiversity and receive subsidies to maintain traditional management. The mixed landscape is of interest for ecological protection because of the mosaic structure formed by forest, marsh and agricultural use (Tierps kommun, 1991). There are 1200 inhabitants in Hållnäs and approximately 70 agricultural enterprises (Tierps kommun, 1991).

Methods

In each country ecological and social data was collected and analysed, initially by country and then in comparative tables and texts. In the first stage, plant diversity data was gathered for the three study sites (section “Comparison of vegetation data”). Secondly, land cover change maps were generated for three key dates in the landscape evolution process (section “Mapping land use change”). Finally semi-structured interviews were conducted with the farmers and landholders managing the landscape to collect their landscape stories, gather their views on land use or abandonment, understand how they manage their businesses and how they plan to develop their activities in the future (section “Social data: interviews”). The results for each stage were integrated and discussed among the project researchers in a seminar (2010) to create a matrix synthesizing the main outcomes and identifying the social and environmental drivers involved (section “Integration of data and production of results”).

Comparison of vegetation data

To get a general understanding of how different land uses have affected flora in the different countries, we compared earlier published data on plant diversity from each study site. Because of different historical and current agricultural management traditions, a perfect match between the land cover/use between countries was not possible. However, we were able to compare the different land covers in terms of the gradient in land use intensity and abandonment of the agricultural use. In Australia, the three key vegetation types of interest were the remnant temperate rainforest (mixture of wet and dry coastal forest types, severely reduced in cover, but considered to have a high value for biodiversity), revegetation (newly established vegetation through planting local indigenous only species), and fields (the cleared landscape, usually grazed and locally denominated paddocks) (cf. Munro et al., 2009). In Portugal, the land uses were agricultural fields (mostly semi-natural pastures and hay fields, but also some crop fields), scrubland and oak forest fragments (cf. Proença and Pereira, 2013). In Sweden, the key land uses were categorized as field, wooded pasture (grazed semi-natural pastures) and coniferous forests (mixed coniferous and deciduous forests used in low intensity forestry) (cf. Lindborg et al., 2008). The number of plant species in each study site is given only as reference, as the field sampling method used to measure species diversity differed between sites. However, the proportion of species occurring in each habitat is informative as well as the proportion of native and exotic plant species.

Mapping land use change

To estimate land use change over time, land use maps were created for each of the three study sites. Three distinct time periods were analysed in the geographic information system ArcMap v.9.1 (ESRI, 2005). The dates were chosen to illustrate key periods in the history of landscape use in each country, and to correlate with the availability of data. In each time period the major land covers were identified, mapped and the area was calculated. These data were then used to build land cover transition matrices for each site.

In the Australian study site (Poowong), pre-European (1840s) vegetation extent was presumed to be 100% cover (KDHS, 1998). Vegetation clearance peaked in approximately 1970, with some revegetation since that time. Revegetation and remnant vegetation were mapped from government department sourced aerial photography. Proportions of different vegetation types in the landscape could be determined for 1970 and 2006.

In both Portugal and Sweden all three time periods used were based on maps and aerial photographs from the last 60 years. In Castro Laboreiro, Portugal, landscape characterization was made through field reconnaissance and photo interpretation for the years 1960, 1990 and 2007 (Rodrigues, 2010). A regressive updating technique was used for the interpretation of the changes in the land use (Bender et al., 2005). The three time layers for the Swedish study case Hållnäs were from 1959, 1979 and 2010. The current and historical land use in Hållnäs was documented through cadastral maps analysing both the current landscape and the landscape from 1950 and 1970.

Social data: interviews

We carried out 1–2 h semi-structured interviews. The contents varied across study sites according to the interests of each research team, but a common set of questions was agreed to provide a common basis for the research presented here. This shared section of the interview inquired about social data across three time zones: 1950s to the present; the present (which was 2010/11); and 2010–2030. Questions regarded socio-economic and demographic data, including quality of life; determining the reasons for present cultivation or the discontinuation of agricultural activity and changes to subsistence/production strategies; perception of landscape change in the last 50 years and services provided by agro-ecosystems; and future perspectives.

Australia

Eleven rural properties from the research area were visited with 18 landholders responding the interviews as individuals, family groups or in partnerships. Ten females and eight males between 30+ and 80+ years participated. Landholders were recruited from a local community-based natural resource management organization (e.g. Landcare) and a community activist association using non-probability convenience and opportunistic sampling. The average size hill farm was 106 ha, which is smaller than the average plains-based South Gippsland dairy farm (138 ha) and northern Victorian dairy farm (160 ha). Some farms have consolidated to grow larger; and this expansion is not necessarily as one single property but rather a series of land titles that are owned or leased around the district, operating as one agribusiness. The vast majority of rural properties were considered farms with only one person viewing their land as a rural residential property, whether or not they derived their main income from the 'farm'. Three interviewees did not classify themselves as farmers and were either retired or employed in non-farming work. All the interviewees resided in Farming Zones as designated in local planning schemes.

Portugal

After a primary informant provided a list of potential respondents, twenty-seven interviews were undertaken. The respondents consisted of 8 men and 19 woman, average age 59 (about 62% of the resident population are women according to the 2011 census). Nearly all of the respondents have resided in Castro Laboreiro for 50 years or more. More than half of the respondents were full time farmers, a third were retired farmers, of which one in three still maintains some type of agricultural activity (often for leisure) and a few of them were part-time farmers with a secondary job. About half of the sample own less than 4 ha of land, and each owner has land scattered across a number of properties (4–6 fields) with an average field size of 0.4–0.8 ha. Of the total sample, almost a half of the households consisted of only 1 person and none of the households had children working in the area. Only 3 respondents were considered large-scale farmers, based on the number of livestock (>50 head of cattle). All respondents were landowners, and only a small fraction additionally rent or work neighbouring private land for free, although most use common land. Income from farming is mainly derived from subsidies for livestock and for maintaining pastures clear of scrubland. None of the respondents acknowledged monetary return from cultivating their land. (Production is for subsistence only.)

Sweden

Nine interviews with small-scale land owners were conducted within the selected study area on Hållnäs peninsula. Farmers were selected from a list compiled by the Swedish Board of Agriculture. The participants are primarily men around 50 years, except one man around 90 years and one female around 50 years. During three of the interviews other family members such as a partner and children were present in the discussion. The size of the properties varied between 5 and 30 hectares of farmland. Land use and farming practices are small scale livestock keeping and hay or ensilage production. These are the typical agricultural activities in the area. Of the nine interviews, a majority of five farmers were leisure or hobby farmers, three were retired and one was a fulltime farmer. Most of the adult inhabitants, including the active farmers, are today working outside the parish or are pensioners. Income

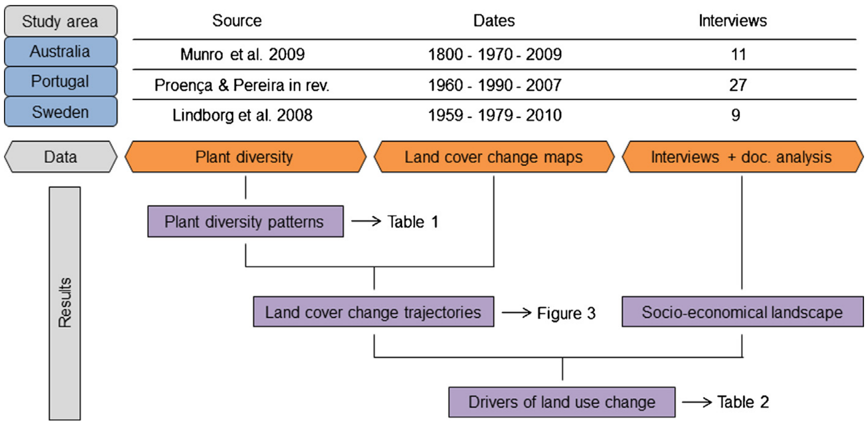

Fig. 2. Methodological framework of the generation and integration of results.

possibilities in conurbations nearby (Skärplinge 10 km) and infrastructure for commuting determines some possibilities for those living and farming in Hållnäs. Some families left Hållnäs completely due to the low income possibilities, although the agricultural tradition and place attachment mean some of the families stay and continue with small-scale farming. Today a number of the inhabitants still have their old hereditary estate and keep the nearest area around the house open with help of sheep or horses but get their main income from districts around Hållnäs.

Integration of data and production of results

A synthesis of the methodological framework can be seen in Fig. 2. The combination of the three types of gathered data (plant diversity, land cover change and social data) was undertaken in a step-wise process of integration. First, the plant diversity inventories were analysed and organized to produce three clusters or types of habitat. This allows for a comparison among the different countries, identifying through them various degrees of agricultural land abandonment in the present day, and their relative importance for floristic diversity. The habitats that active agriculture or abandoned activity produce on each site vary according to the farming decisions and the local conditions (climate, soils, etc.) at each of the sites. Secondly, the data from the plant inventories was used to influence the design of land cover categories identified through mapping land cover change. A quantitative analysis of land cover area for each date at each site provided an understanding of the landscape dynamics that have taken place in each spot over time. The comparison among the different study areas facilitated the identification of similarities and differences across trajectories and time frames. Finally, once the history of the landscape had been understood, and its influence on biodiversity estimated, the integration of the social data gathered from interviews was introduced into the framework, allowing the preparation of a matrix (Table 2). This contained a list of the driving forces that then explained the previous results. The drivers were categorized into three descriptors: pressures, frictions and attractors (Bürgi et al., 2004; Eiter and Potthoff, 2007; Slätmo, 2011). The categories are based on how they affect the ALA locally. Therefore, the categorization emphasizes the importance of context as the same factor or process can have a different effect depending on the local landscape under study. The contents of the matrix and their ratings were determined through the analysis of published historical accounts, socioeconomic trends, government policies, public discourses and community views captured in a variety of official documentation, academic papers, web-based media of relevant organizations and fieldwork interviews with landholders. The matrix was modified during the Australian Workshop (2010) by all three case study teams in a round table meeting. Each country team finalized their individual matrices to reflect the drivers and weightings relative to one another. The matrix distinguishes the strengths of the drivers and the spatial scale at which they operate during two time periods: in the past (1950–2010) and in the future (2010–2030).

Results

Plant diversity patterns

The results emerging from the plant inventories are summarized in Table 1. Sampling methods yielded the identification of 139 plant species in the study site in Portugal whereas in the study sites of Australia and Sweden, almost 250 species were recorded in each. Almost 40% of the identified plants in Australia, however, were exotic species, whereas none of the occurrences in Portugal or Sweden were classified as such. In Poowong, most species of both native and non-native plants were found in revegetation plantings with the lowest biodiversity found in the heavily grazed fields. Castro Laboreiro and Hållnäs had the highest plant species count in agricultural fields and in wooded pastures, respectively. Most agricultural fields in Portugal are pastures and hay fields; and, in contrast with Australia, semi-natural pastures had the highest species diversity. Forests were particularly diverse for plants in Sweden, although in Australia the native plant diversity of forests was also very significant (with 61% of native species occurring in forested areas in Poowong). The least plant rich habitat in Portugal was scrubland, while in Sweden it was fields.

Land use change trajectories

Since the analysis of the land use change trajectories in the Australia study site goes back to pre-European settlement dates, the most dramatic changes to vegetation occur in this case (Fig. 3). Poowong was covered by native vegetation over two hundred years ago, and underwent an intense process of government sponsored clearing. In this region Danish migrant farmers, returned Australian soldiers and other white settlers entered the area from the mid 1870s–1940s (Hartnell, 1974). The peak of clearing was in the 1950s when mechanization assisted in finishing what handsaws and deliberately lit clearing fires had begun. In the 1970s more than 90% of the land was agriculture, comprising introduced and constantly managed pasture grasses (called ‘paddocks’). Even today, only a marginal proportion of the natural vegetation remains and is found along creeks, gullies and roadsides with small patches

Table 1
Comparison of plant diversity across the different habitats in each study site compiled from Munro et al. (2009), Lindborg et al. (2008) and Proença and Pereira (2013).

	Total number of species (native[a])	Habitat	Species (native[a])	Percentage (native[a])
Australia	243 (147)	Temperate rainforest	119 (89)	49% (61%)
		Revegetation	181 (99)	74% (67%)
		Field/grazed field	69 (30)	28% (20%)
Portugal	139	Oak forest	76	54%
		Scrubland	53	38%
		Pasture/crop field	99	71%
Sweden	246	Coniferous forest	179	69%
		Wooded pasture	203	82%
		Field	112	43%

[a] All the species in the study sites in Portugal and Sweden are considered native.

located on private landholdings and in the few existing reserves. Since the 1970s there is a noticeable but weak trend towards decreasing agricultural land uses in this region, and the planting of trees on a 3% cover of previous farmland. Considering that natural vegetation remains steady at 6% of the territory in modern times, this represents a recent increase of 50% in the total forested area over the last 40 years. The low population density leads to the mere 1% of the study area occupied by settlements and other uses, as recorded in Fig. 3.

What can be considered the most intensive type of agriculture in each study case has been less prominent in the European sites throughout this period of time, with less than a 20% occupation by the mid-20th century on both study sites (18% in Castro Laboreiro, 12% in Hållnäs). Yet, these sites have experienced a strong decrease in agricultural area during the last 60 years, with approximately half of the former agricultural land use still in production. In Portugal abandonment began in the mid-20th century and continues to occur (Rodrigues, 2010; Lima, 1996). The characteristically small agricultural fields for pasture or production of rye and potatoes have partly been replaced by scrubland but also, to a minor extent, by Galicio-Portuguese oak forest. This dynamic has allowed scrubland to dominate in the landscape in terms of surface, shifting from a 66% occupation in 1960 to almost 75% nowadays, whereas oak forest has sustained a fairly constant cover over this time period (15–17%). Most of the scrubland on the higher parts of the plateau is still a commons managed by the community to pasture livestock, gather fuel wood, collect plants, and other uses.

In contrast, Sweden has had a decrease both in field and wooded pasture presence as abandonment began at the start of the 20th century, peaking in the 1920s, and by the 1950s the amount of agricultural land had decreased significantly (Ihse, 1995). New wetlands have been established on former wooded pasture areas. As illustrated there is a 6% increase in the category that includes buildings and 'other uses' (e.g. 'wetlands') between 1950 and present time. Forests start to increase their dominance as a land cover from 1950s onwards. Coniferous forest has also increased in the landscape (79% in 2010) due to processes of afforestation on former agricultural land.

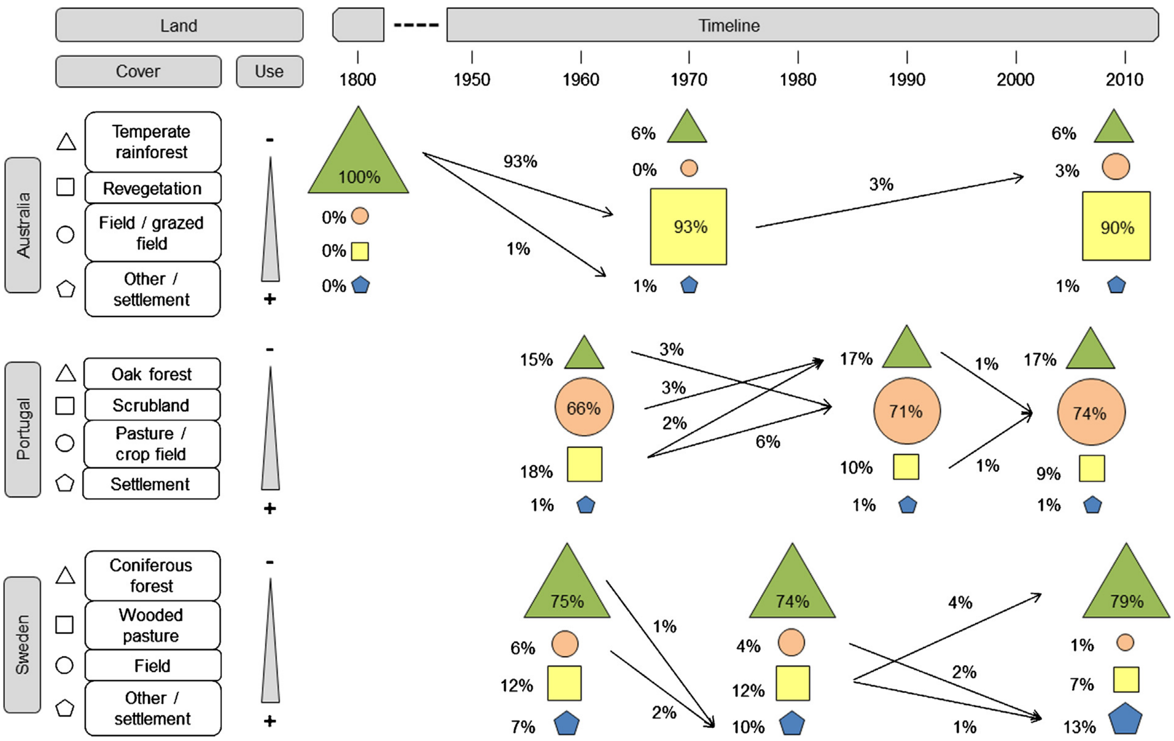

Fig. 3. Trajectories of land use change at each site. Inside or next to each shape is the proportion of land classified in that land use class. Numbers next to the arrows note the percentage of land moving between two different classes in a given time period.

The socio-economical landscape

Interview data provided context for the quantitative findings, in particular clarifying how the socio-economic systems operate at each site.

As a social phenomenon, ALA is least common in Australia, where farmers in the Poowong study area still rely on the long-running activity of dairy farming as a main source of income. Despite a dramatic reduction in the number of businesses in the area during the last three decades of the 20th century (ABS, 1998), the community is still dependant on dairy farming enterprises which are supported by continuing investment in export oriented dairy processing in the region. However, the interviews identified a number of factors currently affecting and transforming this situation: climate change, increasing costs in agricultural inputs, land degradation and volatile milk prices in domestic and global markets. These are creating a fragile operating environment for 21st century dairy farmers. In addition, the ageing of current family farm managers, and a lack of family succession in these agribusinesses, brings the longer term viability of dairying into question. Funding to assist in addressing common natural resource management issues has come from both State and Federal sources over the years and is currently associated with private landowner management of watersheds/catchments and conservation on individual's land (e.g. market-based instruments such as EcoTender). This parallels similar activities for group extension services and community organizations (e.g. Landcare, Greening Australia, Trees on Farms).

In the Portuguese case, changes in the social arena have been coupled with the changes observed in the land cover change analysis. Key influences on the abandonment dynamics include high rates of migration out of rural areas after World War II to secure employment, state policies appropriating common land for afforestation, and Portugal's entrance into the European Union. These increased pressure on local farming enterprises to compete with farms operating at greater economies of scale. ALA is mostly perceived in areas which are characterized by biophysical constraints such as poor soils, steep slopes, harsh climate and isolation. (Castro Laboreiro is located 100 km and extremely winding roads away from the nearest city, Braga.) The traditional mountain farming systems are now facing a collapse. As a consequence of increased agricultural abandonment, scrubland and forest (Galicio-Portuguese oak forest) have increased, and the particular increase of highly flammable shrub areas has escalated the risks of fire occurrence (Moreira et al., 2001b). The main forms of income in the parish are retirement pensions, subsidies for agro-pastoral activities, and most recently, tourism (e.g. lodging and restaurants). Castro Laboreiro is a European Union Less Favoured Area (an area where agricultural production or activity is more difficult because of natural handicaps), and therefore farmers benefit from special subsidies. Furthermore, the landscape mosaic is classed as being of high nature value farmland and farmers can benefit from agri-environmental payments aimed at maintaining such habitats. Meanwhile, the potential benefits of oak forest regeneration to biodiversity and local ecosystem services are not acknowledged by the current policies, which are mainly focused on the prevention of ALA.

Biophysical conditions have had an influence on the human activities in the Swedish case area and their viability over time. Agriculture has low productivity here and living solely on farming has traditionally been difficult due to harsh climate, poor soils and small parcels. Farming activities have declined since the 1950s and today, primarily livestock keeping and hay production remain. The majority of the population does not have a working farm and those who have are mainly 'leisure farmers', keeping the land open while working outside the parish for an income. This situation indicates that the landowners in Hållnäs have other motives for decisions about land use than landowners who survive on farming. Challenges for the future of agricultural activity in Hållnäs relate to the surplus of agricultural land, from a European Union perspective; while small units and relatively poor soil quality make it difficult to develop economically viable agricultural businesses in the parish. This makes alternative working opportunities crucial for continued farming. Workplaces within commuting distance are therefore essential in order to keep active the small-scale farming lifestyle as a part-time or hobby activity. On the other hand, subsidies are aiding small farm units, opportunities arise in the field of summer tourism and leisure residents are believed to positively influence the future sustainability of the community, culminating in the preference for a mosaic landscape (Stenseke, 2009).

Drivers of land use change

The drivers that have been identified for ALA for the various areas were then categorized into pressures, frictions and attractors (Table 2). Triggers for ALA (for example, the post WWII reconstruction that led to a rural exodus from Castro Laboreiro, before the timeline of this study) undoubtedly manifest as pressures or frictions within other time frames (as apparent in the ongoing decline in population).

Ongoing pressures from global market forces have been important in all three study sites, emphasizing the impact of international scale on local outcomes. The three areas are somewhat aligned with regard to historical pressures related to the expansion of state-owned forestry and a sense of remoteness, driven mainly at the local level. Remoteness, surprisingly, does not seem equally correlated with a relative decline of infrastructural accessibility, or the relative decline of working opportunities, or a diminishing level of public and commercial service for the three study areas. These factors are only important for future agricultural land use in Castro Laboreiro and Hållnäs. In Poowong pressures of reforestation and a sense of remoteness have historically had some influence on land use change dynamics locally, but these will be less relevant in the future as the area experiences the influence of urbanization pressures from an expanding metropolitan population. In contrast, in Castro Laboreiro and Hållnäs re-afforestation and remoteness have and are contributing significantly to ALA. Nature conservation regulations promoted at the local level are gaining momentum in Australia, but they remain a pressure for agricultural land abandonment in the Portuguese case study and less prominently in the Swedish case. It is also important to note that some pressures interact, creating positive feedback. For instance, the relative decline of working opportunities leads to out-migration which leads to a diminishing level of public and commercial service which in turns leads to further migration and further decreases in the level of public and commercial services (Figueiredo and Pereira, 2011).

In the past, frictions preventing the abandonment of agricultural land have appeared or have been put in place in the three study areas, mainly from the national level, but some of them are increasingly coming from the local governments and communities. In the future, however, the general trend indicates that there will be an increase in the influence of such driving forces, but not in relation to community cohesiveness and farming identity among land owners (i.e. the degree to which citizens are committed to the prosperity of the community and to which land owners are committed to ongoing farming). These latter are expected to act less as a friction to ALA in Portugal and Sweden, but is likely to remain an important social driver in Australia. State and regional campaigns for rural development will also be important local frictions in Australia, whereas in Europe though equally determining, emanate from the national level. Official funding and subsidies for land management, tourism and second home owners represent more obvious differences, as frictions within the study areas. Their influence has been and will be limited in the Australian case. They have been important drivers

Table 2
Drivers of land use change identified in the study areas along two time periods (past: 1950–2010, future: 2010–2030). More influential drivers are given a higher number of + (from 1 to 3) and 0 for nil influence, whereas the scale of their influence is divided into three levels according to the classification: I = international, N = national, and L = local.

Driving forces to agricultural land use change		Of specific importance in Poowong		Of specific importance in Castro Laboreiro		Of specific importance in Hällnäs	
		Past	Future	Past	Future	Past	Future
Pressures to abandon agricultural practices on the land	Market driven changes in economic conditions for farming	+ I	++ I	++ I	+++ I	+++I/L	+ I/L
	Expansion of forest land owned by companies/state	++L	+ L	++N	0 L/N	++ L	+ L
	Diminishing level of public and commercial service	0	0	++L	++ L	+ L	++ L
	Relative decline of working opportunities	+ L	0	+++L	+++ L	+++ L	++ L
	Relative decline of infrastructural accessibility	+ L	0	++L	+ L	++ L	+++ L
	Increased nature conservation regulations	0	++ L	+ N	+ N	++ N	+ N
	Feeling of continuing remoteness	++L	+ L	++L	++ L	+ L	+ L
Frictions to abandon agricultural practices on land	Cohesiveness and the farming identity among land owners	++L	++ L	+++L	+ L	++ L	+ L
	Official funding and subsidies for land management	+ N	+ N	++N	+++ N	++ N	+++ N
	State campaigns for rural development	++L	++ L	+ N	++ N	+ N	++ N
	Tourism and secondary home owners	0	+ N/L	+N/L	++N/L	+ N/L	+++N/L
	Appreciation of natural and cultural values	+ L	++N/L	+ N	++ N	+ N	++ N
Attractors for abandonment of agricultural practices on the land	Physical conditions for cultivating	0	0	+++L	+++ L	+++ L	+ L
	Physical conditions for livestock keeping	+ L	0	+++L	++ L	+ L	0

in the past in the Portuguese and Swedish areas. Further, they are expected to play an even more prominent role there in preventing ALA in the future.

Attractors for abandonment include the physical conditions in the Swedish and the Portuguese cases areas, which are less favourable, and that has underpinned their abandonment. In northern Portugal, this is likely to continue, while in Hällnäs, the development has gone so far, that the remaining lands are now, in the main, managed by leisure farmers, and there are therefore, other motives than that of economic revenues keeping them in agricultural practice for the future. In Poowong, there are currently favourable climatic and soil conditions for commercial cattle grazing (understood as a complementary use to dairying); therefore, the biophysical conditions are not expected to attract considerable ALA.

Discussion

Comparison of the three study cases

Comparative studies offer the opportunity to analyse similarities and differences between study areas (George and Bennett, 2005). The integrative results provide a global connectivity; contextualizing the findings in an interesting and more useful way than would be the case if they had remained within the boundaries of their own disciplines (Redman et al., 2004). In this case, there is opportunity to compare landscape trajectories in three areas that, due to their diverse biophysical features, but not least due to their particular socioeconomic and cultural environments, have undergone substantially different transformations over time. Swedish and Portuguese agricultural landscapes have been cultivated over a much longer time than the Australian landscapes (see section "Land use change trajectories"; Meeus et al., 1990; Powell, 1996). This is clearly noticeable when dynamics of agricultural land abandonment are contrasted. Whereas the proportions of land uses in the landscape in the European study cases seem relatively stable over the period of time for which data has been recorded, dramatic land use change has occurred in the Australian case study over a somewhat longer period of time, despite a shorter period of cultivation overall. The rural areas of Castro Laboreiro and Hällnäs are the result of a much longer history of profound agricultural landscape management.

In Europe social drivers have played an important role in stimulating modifications to these landscapes. A number of local decision factors need to be considered as farmers (as the main stakeholders responsible for the modification of the agricultural landscape) organize land management. Albeit to very different degrees, all three study areas have attracted some abandonment of cultivation or livestock keeping due to challenging physical conditions. In the last 60 years, factors like the sense of remoteness or the decline of infrastructural accessibility and working opportunities associated with their peripheral locations have pressed many farmers to give up farming, abandoning their cultivation or grazing. Finally, frictions coming from the local or the national level such as those associated with maintaining or stimulating cohesiveness and the farming identity among land owners through direct or indirect state campaigns for rural development (sometimes coupled with official funding or subsidies) have attempted to counter ALA. In effect, mainly in Europe, attention has turned to emphasizing an appreciation of natural and cultural values across the case sites. The development of conservation values across formerly production landscapes has played a part in preventing some abandonment, or at least diminishing its prominence, due to subsidies, land stewardship programs, and agri-tourism initiatives.

In Table 2 we have synthesized the driving forces of agricultural land use change, offering an estimation of their relative importance towards land use change and noticing the scale at which they occur. When these results are compared to the landscape trajectories presented in the section "Land use change trajectories", a correlation between the intensity of land use change in the last decades and the relative importance of the driving forces becomes apparent. In the last 40 years, only a 3% of agricultural land has reverted to forested areas in Poowong (mostly framed as being integrated into the farming system/lifestyle of the property), whereas in a slightly longer period of time, the changes indicating ALA in Castro Laboreiro and Hällnäs account for approximately 10% in absolute terms. In this same time scale, pressures and attractors of abandonment have been deemed to be of much less importance in the Australian case study, whereas the matrix reveals a much stronger effect of such drivers in the Swedish and Portuguese sites. Again, physical conditions play a role in determining such dynamics. Poowong offers the most suitable biophysical conditions for farming; but most of its drivers are social in origin. Excluding market driven changes in economic conditions for farming, which affect the three study sites with similar intensity as WTO members, yet with different effects (Dibden et al., 2009; Hamblin, 2009), all other pressures have been stronger in Europe. Furthermore, both isolation (relative to local markets, access to infrastructure and services) and ageing demographics are also more of an issue in Sweden and especially in Portugal than they are in the Australian case. Nonetheless, if the differences in the mapped trends of ALA are not even more marked, this might be due to the fact that frictions preventing the abandonment of agricultural practices have also been stronger in the European sites than they have been in Oceania. This might be related to the fact that in the Swedish and Portuguese case studies, a multifunctional rural transition has provided an increased balance between the economic, the social and the environmental capital, resulting in what Wilson (2010) characterizes as a strong multifunctionality leading to a reinforcement of the resilience of these local communities in a post-productivist era. The multi-functionality of mosaic rural landscapes made of cropping fields or semi-natural grasslands, hedgerows or patches of trees and masses of tree cover has set the basis for national support plans and subsidies, state campaigns for rural development and an increased appreciation and interest in preserving and enjoying their recreational values. This is evidenced by Swedish farmers, albeit uneconomically, who keep managing their lands for their aesthetic value, or when the heirs of emigrants from Castro Laboreiro locate their second home in the parish, or with the emergence of agri-tourism in these same areas (Granvik et al., 2012). By contrast, in Australia there have been several State Government attempts to support farmers to abandon agriculture on the initially cleared eastern slopes of the Strzelecki Bioregion for tree plantations, and afforestation has been promoted in the pursuit of environmental enhancements such as erosion prevention and hydrology management (Cary and Roberts, 2011). Even if their success has been limited (only a 3% of the land has changed use from agriculture to trees in 40 years), it contributes to biodiversity conservation as it matures. Therefore, the landscape in the Australian case study exhibits a rather weak multifunctionality, in which commodity farming, called 'productivist agriculture' by Holmes (2006), and simply, 'production' landscapes by Barr (2005), dominate social and environmental concerns.

Integrating ecological and sociological data

By integrating sociological data with plant inventories generated in the study areas we are able to link some of the processes occurring on the landscape and related to agricultural land abandonment with the impacts that these dynamics might have on biodiversity. The climate and geomorphologic conditions, however, differ between areas hence it is difficult to draw strong conclusions based on the observed differences in the richness and diversity of plant species among study sites. However, within sites, changes associated with altered management can be examined. Intermediate intensified agriculture often holds the highest biodiversity compared to highly intensified or abandoned areas (Tscharntke et al., 2005). Difference within countries shows that management intensity partly explains differences in species richness as 'field' in Portugal and 'wooded pasture' in Sweden (both representing intermediate intensity) had the highest richness compared to revegetated areas in Australia. Specific for the Australian case is the distinction between native and exotic plants, where the longer history of landscape transformation occurring in the European study sites has made this distinction much more difficult for the natural sciences (Kornaš, 1990). Much of the Australian biodiversity, and even more so the native one, is recorded in the remnants of original forests (89 out of 147 native species are present). Even if such remnants account for only 6% of the study area, these figures prove how valuable these patches are for the conservation of native flora. Land abandonment processes in Portugal and Sweden are further strengthening the presence of land uses that are already dominant in the landscape, as scrubland in Castro Laboreiro (74% in 2007) and coniferous forest in Hållnäs (79% in 2010). This diminishes diversity in the landscape, and is detrimental to plant species which occur in agricultural fields and pastures.

The distinction of the situation in Australia and the situation in Portugal and Sweden leads to an important observation. In the European case studies, social forces are driving the landscape towards land abandonment processes that by reducing its diversity of uses are diminishing its potential value for biodiversity associated with high-natural value farmland. In Australia, factors facilitating further abandonment of fields and promoting their transition towards the creation of forested habitat would increase opportunities for species to establish themselves in the rural landscape. Thus, current ecological communities would be strengthened by not only increasing the total habitat availability but also through an improvement of its overall connectivity (Fahrig et al., 2011). The policies in Portugal and Sweden are transitioning their rural landscapes from just agricultural production systems to multifunctional systems. Even if their attractors for ALA are stronger than in Australia and their pressures to abandon land are deemed more influential, prominent frictions exist or have been put in place at the local or national levels to prevent such undesirable change, with support from most of these communities (Renwick et al., 2013). By contrast, in Australia, where a certain degree of ALA would be beneficial for biodiversity if adequately managed, pressures and attractors are less influential, and some frictions, orientated towards sustaining the intensification of agricultural land uses to ultimately increase commercial productivity, are preventing this change from taking place. This makes farming activity in Australia very dependent on its capacity to produce food to satisfy exports markets; and at the same time prevents indigenous species from inhabiting larger, less fragmented habitat, reducing the functional resilience of both entities to eventual changes. In Europe, due to a somewhat larger equity of land uses in the landscape and especially due to a number of frictions preventing sudden ALA from occurring, the stability of the traditional agricultural landscape and hence the resilience of the species, populations and communities inhabiting it, is increased. Yet, long term processes of ALA, and the threats they pose on biodiversity, have not been completely prevented either in Sweden or Portugal.

A scalar analysis of these dynamics also provides some interesting insights to explain the substantially different outcomes in relatively similar study areas. In fact, one could argue that, following a conception of scale based on size considerations only, and operating as pre-given nested hierarchical levels, the three study areas and the processes taking place in them look very similar indeed (Jonas, 2006). In this sense, each covers a similar size of territory, produces a similar type of commodity for which global demand and markets exist, is regulated according to the policies of a state, and is managed at the local level by a group of struggling farmers who see how their average age increases while the number of viable farm businesses decreases. This description fits comfortably with the germinal framework envisaged by Taylor (1982), building on Wallerstein's world systems theory, characterized by the presence of a global scale at which the world economy operates, a nation state scale at which ideology is expressed and, finally, a local scale where daily experience happens and individuals can feel the consequences of arrangements coming from the superior scales. For a long time, this three-level framework has heavily influenced the widespread vision that the important decisions on higher levels in the hierarchy are the ones with higher causal power to explain processes observed at the lower level (Marston et al., 2005). Furthermore, it has contributed to the conception of a duality between a global level which imposes its preferences – usually of an economical nature – and a local level which receives its negative consequences and therefore struggles to resist and to fight them (Howitt, 2003).

Table 2, where for practical reasons we have adopted the simplified view of three levels of organization influencing farmers' decisions, indicates differences coming not only from the different biophysical realities of the landscape in which each group of farmers operate, but also from their reaction to – or perception of – similar, if not virtually identical, drivers. After all, conceiving scale from a constructivist approach, in which relationships between social groups, and not size, define the different levels, the interpretation of our findings changes dramatically (Swyngedouw, 1997; Moore, 2008). Under this scope, and observing the alignment between the behaviours and perceptions of Poowong farmers and export markets demands for competitive production, it is clear that relationships in the Australian case study operate, mostly, at a global level. Even if other attractors, pressures and frictions appear in our model, the global scale at which these farmers operate seems highly influential in explaining the dynamics that have been observed in the landscape, its land use changes and the problems that, ultimately, entail for some species, habitats and soils. In this case, there is a strong connection between the global and the local, with the latter significantly contributing to build the former, as well as subject to its influence (Massey, 1994). Global markets also push farmers in European agricultural areas to be competitive in terms of production or otherwise to cease their activity and abandon the land. Moreover, local specificities like remoteness or poor soils add further pressure for ALA in the study areas of Castro Laboreiro and Hållnäs. The frictions that the European Union, the respective states and the cultural traditions of the interviewees have put in place, however, substantially change the relationship that farmers in these local areas have with other scales and groups. Subsidies and lifestyle tradition act as economic and cultural firewalls not just against some global drivers, but to local constraints too. At the same time, these farmers are more connected to other social groups, providing a preferred landscape for aesthetics, tourism and second homes (which one could argue to be global trends, in the global North at least). Ultimately, their multifunctional response and their landscapes, showcases more complex scalar links than that of their Australian counterparts. Therefore, it is much more difficult to provide a label for the scale at which farmers in Castro Laboreiro and Hållnäs operate, even if, in terms of constructed relationships, it is clearly closer to the local level than that of farmers in Poowong. In the European sites, when analysing ALA and its impacts on the landscape and biodiversity over time, the identification of local (and regional) scales as a level of resistance against global forces remains, explaining some of the local land use change dynamics observed. Such scalar considerations and differences need to be taken into account if policies regarding ALA and its drivers are to be implemented and successful management is to be achieved in the future.

A look into the future

Driving forces steering agricultural land use change in the three study areas are not expected to change substantially during the next 20 years. Nonetheless, our results offer some interesting insights into the socio-economic dynamics expected in the respective landscapes.

Pressures to abandon agricultural practices on the Australian land will tend to increase due to both further competitiveness demands from global markets that will make some businesses unviable and regulations for nature conservation. On the other hand, however, the relative proximity of the Poowong study area to the metropolitan area of Melbourne is likely to affirm pressures related to spatial isolation easing, as the expansion of the city leads to commuters seeking cheaper housing ex-Melbourne. This provides opportunities for the development of stronger local markets, would reduce the sense of remoteness, increase infrastructural accessibility and offer more working opportunities. Isolation and remoteness are factors expected to keep driving ALA in Hållnäs and Castro Laboreiro, as their location will continue to hinder working opportunities, infrastructural development and service provision. New communication technologies can reduce virtual distances to larger cities in the future. This effect, valid for Australia and Sweden, will be limited in the short term in Portugal due to a somewhat older population, less prone to adaptation to new technologies.

Frictions to abandon agricultural activity are generally expected to increase in future years, mainly due to the development of alternatives other than farming, such as tourism, second homes, or simply the provision of funds and subsidies for land management. Such a change, however, will also be associated in Hållnäs and Castro Laboreiro with a loss in the cohesiveness and farming identity among land owners, a factor that, until now, has played an important role in resisting further agricultural activity decreases and land abandonment. Attractors for abandonment of agricultural practices will remain stable in Australia, but will generally diminish in Portugal and Sweden, as it is expected that, after experiencing processes of ALA for long periods of time, lands remaining cultivated and grazed will already be the most valuable ones, yield-wise or for their provision of ecological services and scenic views.

The effects of the described changes of social drivers over the agricultural landscapes indicate that no dramatic changes are to be expected in the proportions of land uses in the Poowong study area. Scenarios for Portugal and Sweden are much more uncertain, since, resistance to land abandonment posed by reinforced frictions and weakened attractors are to be confronted by some mounting pressures, while others are expected to become less dominant. Even if weakly, the overall balance would indicate that ALA in the European sites is likely to recede in the next twenty years, so even if further abandonment can still be envisaged, the trend points towards a stabilization. Should this happen, the proportions of forests, pastures and fields in the study areas of Hållnäs and Castro Laboreiro would not be significantly different from present day. The existence of feedback loops renders the dynamics of ALA harder to reverse as well.

Despite the different characteristics of the study areas and changes in the social drivers that are commanding the process of ALA with various degrees of intensity and different time scales in the respective study areas, the stabilization.of agricultural land use area would bring little implications for biodiversity. However, just small modifications in some of the dynamics, such as ALA eventually gaining a little momentum in Australia could open a new window to increase forested land cover and with it significantly increase the availability of habitat for multiple species of flora and fauna, while increasing the resilience of the natural system. Additionally, the restoration of tree cover on some of the steepest slopes currently cleared for dairy farming, would limit environmental problems such as tunnel and gully erosion or water pollution. Further ALA in Sweden or Portugal could have exactly the opposite effect, since the traditional agricultural land uses already remain in relatively small proportions, and natural succession would bring forth more homogenous forest cover, threatening the provision of some ecosystem and cultural services that these landscapes satisfy. Rewilding, while contested, promotes the reintroduction of large herbivores and is another approach to maintaining open landscapes associated with biodiversity (Navarro and Pereira, 2012).

Policies implemented in the latest years have impacted ALA dynamics in each of the three study areas. In industrialized European countries, the restructuring of the agricultural sector has generated 'uneven spatial consequences' (Potter and Tilzey, 2005, p. 595), creating a two pathway narrative where productivist agricultural spaces are promoted at the same time as governments support a different version of the 'consumption countryside' (Potter and Tilzey, 2005, p. 596). This approach is focused on multi-functionality and promotion of Geographical Indications (GIs) – like *terroir* – which act to protect 'rural communities, farming systems and landscapes against the full rigours of neo-liberalism' (Dibden et al., 2009, pp. 306–7). In the Australian case, government policies to encourage 'marginal' farmers and landscapes out of production have encountered complex social issues such that the poorest farmers may be too poor to be able to leave, or have too few options if they do (Sysak, 2013). In general federal agri-environmental policies in Australia can be seen as WTO compliant (Dibden et al., 2009) even as the last five years have also witnessed a confused social agenda in relation to drought and energy policy (Beilin et al., 2012). On the other hand, the European Union seems to be firmly engaged in promoting its current agenda for the foreseeable future, and it is therefore expected that Castro Laboreiro and Hållnäs will be able to rely on the stability of policies promoting their multi-functionality (van Berkel and Verburg, 2011). This factor should help to avoid further ALA in these European locations. The impact of future policies in Australia, however, is much less clear, despite the realization that their impact could be much more dramatic. Not however, while policy focuses the landscape is on production (Hamblin, 2009).

In considering landscape dynamics in the categorisation of driving forces, it is evident that very different measures could develop in each study site, adapted to their specific goals and biophysical characteristics. The outcomes can be obtained in a range of different ways to increase or reduce pressures and frictions where desirable, acknowledging that increasing pressures (or just letting them build up through inaction) will certainly lead to further ALA, while the opposite holds true regarding frictions. No overarching influence can achieve such diverse results or deliver desirable outcomes everywhere. Therefore, forces acting at the international scale, such as global markets, are poorly suited to steer more complex changes – e.g., multifunctionality – in Poowong, Castro Laboreiro and Hållnäs. On the contrary, national and local scale actions, tailored to suit each reality and set of expectations will be important.

Conclusion

Agricultural land abandonment is at one end of a continuum of land use change that has transformed rural landscapes in the study areas of Portugal and Sweden. It is at a much earlier stage with potential to increase in the Australian case. The generation of historical land use cartography and the analysis of the trajectories of land cover change support this conclusion. We have identified a set of driving forces, classified into pressures, frictions and attractors that clarify why ALA, noting its temporal and spatial scales, occurs differently in each study area. Pressures and attractors pushing for agricultural activity abandonment are stronger in Portugal and Sweden than in Australia. Generally more (institutionalized) frictions are in place in the European sites in order to prevent further change. Across the countries, a strong feeling of cohesiveness and farming identity has slowed further ALA. In the future other influences might provide new opportunities for their management (e.g. tourism, affordable living, ecological services or appreciation of natural and cultural values). This shift is observed as underway in Sweden, evolving in Portugal and nascent in Australia. These factors have consequences for biodiversity.

Implications for biodiversity vary, as ALA is not per se a positive or negative process. Social drivers can be aligned or misaligned to steer landscape change in a desirable direction, to favour or through an inadequate path, to reduce the endurance of both the farming system and the natural system. Planners and policy makers can evaluate the current ALA situation in each area, assessing the social drivers that determine its dynamics, redefining policy goals, strategically aligning land use zoning to support agro-ecological systems, introducing market-based instruments to support land stewardship and setting a socially responsive agenda. We conclude that in Poowong, the stimulation of driving forces promoting a well-managed abandonment of some cleared areas and their transition towards the protection of remnant forest patches, the plantation of new ones and the revegetation of networks would be highly beneficial for Australian biodiversity, minimally disruptive for current dairy farming operations and would bring opportunities for alternative types of rural development. In Portugal, ALA tends to homogenize the landscape at the local level, decreasing the abundance of species currently associated with high-natural value farmland and grassland and increasing wildfire risks. Remoteness and an ageing population are challenges to overcome if further abandonment is to be avoided in Castro Laboreiro. In the future new opportunities are emerging in Portugal regarding alternative uses of the traditional agricultural landscape, making it desirable both for human and floristic communities. A similar dynamic is occurring in Hållnäs, Sweden, where ALA is already perceived as a threat to biodiverse and socially responsive landscapes. Because of this, there is a strong governmental support to preserve the ecological value of the traditional landscape.

Finally, we provide a reflection on the methods for integrating a three country study among various disciplines to better understand ALA. Aside from the practical variations associated with land management regimes, diverse cultural expectations and historical record keeping, was the everyday reality of the same descriptive words having different meaning in each country. For example, in Portugal, 'pasture' is untended except by the animals, whereas in Australia it is closely managed with seeding and fertilizing. As a consequence of the complexity inherent in such international comparisons, we have benefitted from the integration of qualitative and quantitative data. We intend that others may build on our presentation of data to extend the ability of learning from and with each other in ways that edify what seem to be particularly local issues until connected to similar experiences elsewhere. As such, collaboration for interdisciplinary research is very much about finding processes for transforming evidence into meaningful and useful analysis.

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C. Appendix: Supplementary Material to chapter 2.

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Supplementary Table 1.

List of species identified as benefiting from land abandonment and rewilding

(expanded from Russo, 2006).

A. Invertebrates

Species	Vernacular	Location of the study	Ref.
Abax ater			1
Abida polyodon			2
Allolobophora multali			2
Aphaenogaster subterranea			3
Camponotus lateralis			3
Candidula gigaxii			2
Chondrina avenacea			2
Cochlostoma septemspirale			2
Crematogaster scutellaris			3
Dendrobaena mammalis			2
Formica gerardi			3
Glomeris marginata	Pill milipede		4
Leptothorax racovitzai			3
Lumbricus castaneus			2
Pagodulina pagodula			2
Plagiolepis pygmaea			3
Pomatias elegans			2
Porcellio gallicus			4
Punctum pygmaeum			2
Solatopupa similis			2
Solenopsis fugax			3
Steropus madidus			1
Urticicola glabellus			2
Vitrea contracta			2
Xerosecta cespitum			2
Zebrina detrita			2

B. Aves

Species	Vernacular	Location of the study	Ref.
Aegithalos caudatus	Long-tailed tit	NW Mediterranean	5
Anthropoides virgo	Demoiselle crane	Steppe, Russia	6
Anthus trivialis (**)	Tree pipit	NW Mediterranean	5
Aquila chrysaetos (**)	Golden eagle	Lithuania	7
Aquila heliacal	Eastern imperial eagle	Steppe, Russia	6
Aquila nipalensis	Steppe eagle	Steppe, Russia	6
Aquila pomarina	Lesser spotted eagle		8
Carduelis cannabina (*)(**)	Eurasian linnet	NW Mediterranean	5
Carduelis carduelis (*)(**)	European goldfinch	NW Mediterranean	5
Carduelis chloris (*)(**)	European greenfinch	NW Mediterranean	5
Certhia brachydactyla	Short-toed treecreeper	Apennines, Italy; NW Mediterranean	5; 9
Certhia familiaris	Eurasian treecreeper	Apennines, Italy	9
Circus pygargus	Montagu's harrier		8
Columba palumbus (*)	Wood pigeon	Andalusia; NW Mediterranean	10; 11; 5
Corvus corone (*)	Carrion crow	NW Mediterranean	5
Crex crex (**)	Corncrake	Lithuania	7
Cuculus canorus (*)	Common cuckoo	NW Mediterranean	5
Dendrocopos major	Great spotted woodpecker	NW Mediterranean; Apennines, Italy	5; 9
Dendrocopos minor	Lesser spotted woodpecker	NW Mediterranean	5
Dryocopus martius	Black woodpecker	NW Mediterranean	5
Emberiza cirlus (*)	Cirl bunting	NW Mediterranean	5
Erithacus rubecula (*)	European robin	Alps, Apennines, Italy; NW Mediterranean	12; 5; 9
Fringilla coelebs (*)	Eurasian chaffinch	Apennines, Italy; NW Mediterranean	11; 5 ; 9
Galerida theklae (*)	Thekla lark	NW Mediterranean	5
Haliaeetus albicilla	White-tailed eagle		14
Hippolais polyglotta (*)	Melodious warbler	NW Mediterranean	11; 5
Jynx torquilla	Eurasian wryneck	NW Mediterranean	5
Lanius collurio (*)(**)	Red-backed shrike	NW Mediterranean	5
Lullula arborea (*)	Wood lark	NW Mediterranean	5
Melanocorypha calandra (*)	Calandra Lark	NW Mediterranean	5
Miliaria calandra (*)(**)	Corn bunting	NW Mediterranean	5
Parus ater	Coal tit	Alps, Italy	12; 5
Parus caeruleus	Blue tit	Apennines, Italy; NW Mediterranean	11; 5; 9
Parus cristatus	Crested tit	NW Mediterranean	11; 5
Parus major (*)	Great tit	NW Mediterranean	11; 5
Parus montanus	Willow tit	Alps, Italy	12
Parus palustri	Marsh tit	Apennines, Italy; NW Mediterranean	5; 9
Phasianus colchicus	Common pheasant	Apennines, Italy	9

Species	Vernacular	Location of the study	Ref.
Phoenicurus phoenicurus	Common redstart	Apennines, Italy	9
Phylloscopus bonelli (*)	Bonelli's warbler	NW Mediterranean	5
Phylloscopus collybita (**)	Chiffchaff	Alps, Italy; NW Mediterranean	12; 5
Picus viridis (*)	Eurasian green woodpecker	NW Mediterranean	5
Prunella modularis	Dunnock	Alps, Italy; NW Mediterranean	12; 5
Regulus ignicapillus	Firecrest	NW Mediterranean	11; 5
Regulus regulus	Goldcrest	Apennines, Italy; NW Mediterranean	5; 9
Saxicola torquatus (*)	Common stonechat	NW Mediterranean	5
Serinus citrinella (*)	Alpine citril finch	NW Mediterranean	5
Sitta europaea	Wood nuthatch	Apennines, Italy; NW Mediterranean	5; 9
Streptopelia turtur (**)	European tuttle-dove	Apennines, Italy	9
Sylvia atricapilla (*)	Blackcap	Apennines, Italy; NW Mediterranean	11; 5; 9
Sylvia borin (**)	Garden warbler	Alps, Italy	12
Sylvia cantillans	Subalpine warbler	Apennines, Italy; NW Mediterranean	5; 9
Sylvia curruca	Lesser white-throat	Alps, Italy	12
Sylvia hortensis (*)	Orphean warbler	NW Mediterranean	5
Sylvia melanocephala (*)	Sardinian warbler	NW Mediterranean	11; 5
Sylvia undata (*)(**)	Dartford warbler	NW Mediterranean	5
Tetrax tetrax (**)	Little bustard	Steppe, Eastern Europe	13
Troglodytes troglodytes (**)	Winter wren	Alps, Italy	12
Turdus merula (*)	Eurasian blackbird	NW Mediterranean	5
Turdus philomelos (*)	Song thrush	Apennines, Italy; NW Mediterranean	5; 9

C. Mammalia

Species	Vernacular	Location of the study	Ref.
Alces alces	Moose		14
Arvicola terrestris	European water vole	Jura mountain, France	2
Bison bonasus	European bison	Eastern Europe	15; 14
Canis lupus	Grey wolf		16; 17; 18; 19
Capra aegagrus hircus	Feral goat	Aragon, Spain	17
Capra pyrenaica	Iberian ibex	Andalusia; Aragon, Spain	10; 17
Capreolus capreolus	Roe deer	Andalusia; Aragon, Spain	10; 17
Castor fiber	Beaver		7; 14
Cervus Elaphus	Red deer	Andalusia; Aragon, Spain	10; 17; 14
Dama dama	Fallow deer	Aragon, Spain	17
Genetta genetta	Common genet	Doñana NP, Spain	2

Species	Vernacular	Location of the study	Ref.
Gulo gulo	Wolverine	Northern Scandinavia	16
Hystrix cristata	Crested porcupine	Southern Europe	2
Lutra lutra	Otter		7
Lynx lynx	Eurasian lynx		16; 20; 14
Microtus arvalis	Common vole	Jura mountain, France	2
Muscardinus avellanarius	Hazel dormouse		2
Ovis ammon	Argali	Aragon, Spain	17
Rupicapra pyrenaica	Pyrenean chamois	Aragon, Spain	17
Saiga tatarica	Saiga antelope	Steppe, Russia	6
Sciurus vulgaris	Eurasian red squirrel		2
Sus scrofa	Wild boar	Andalusia; Aragon, Spain	10; 17
Ursus arctos	Brown bear		16; 17; 21
Vulpes vulpes	Red fox	Doñana NP, Spain	2

(*) Despite reporting increasing trends, these species rely on open land, scrubland and intermediate woody vegetation.

(**) Species that are also considered as being negatively affected by land abandonment and rewilding in some locations.

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Supplementary Table 2.

List of species identified as being negatively affected by land abandonment and rewilding (expanded from Russo, 2006).

A. Invertebrates

Species	Vernacular	Location of the study	Ref.
Allolobophora icterica (*)			1
Amara brevicollis			1
Amara communis			1
Amara lumicollis			1
Aphaenogaster senilis (*)			1
Aporrectodea caliginosa			1
Cataglyphis piliscapus			1
Chortippus parallelus			1
Cilindroiulus caeruleocinctus (*)			1
Decticus verrucivorus	Wart-biter		1
Euthystira brachyptera			1
Formica cunicularia			1
Formica rufibarbis			1
Glomeris annulata (*)			1
Lasius niger	Black garden ant		1
Leptoiulus belgicus (*)			1
Lumbricus terrestris			1
Messor barbarus			1
Messor scabrinodis			1
Ommatoiulus rutilans			1
Tetramorium caespitum (*)	Pavement ant		1
Tetramorium ruginode			1
Tetramorium semilaeve			1
Trochoidea cylindrica			1
Trochoidea geyeri			1

B. Aves

Species	Vernacular	Location of the study	Ref.
Acrocephalus paludicola	Aquatic warbler		1
Alauda arvensis	Skylark	NW Mediterranean	1; 2
Alectoris rufa	Red partridge	Picos de Europa, Spain	3; 2
Anthus campestris	Tawny pipit		2
Aquila adalberti	Spanish imperial eagle		1
Bonasa bonasia	Hazel grouse	Haut Jura, France	3
Bubo bubo	Eurasian Eagle-owl		1
Bubulcus ibis	Cattle egret		1
Calandrella brachydactyla	Greater short-toed lark	NW Mediterranean	1; 2
Carduelis cannabina	Linnet		1
Clamator gladarius	Great spotted cuckoo	NW Mediterranean	2
Corvus corax	Common raven		4
Corvus monedula	Jackdaw		4
Coturnix coturnix	Common quail	NW Mediterranean	2
Emberiza cia	Rock bunting	NW Mediterranean	2
Emberiza citrinella	Yellowhammer	NW Mediterranean	2
Emberiza hortulana	Ortolan bunting	NW Mediterranean	2; 4
Falco naumanni	Lesser kestrel	Portugal	5; 1
Galerida cristata	Crested lark	NW Mediterranean	2
Garrulus glandarius	Eurasiam jay	NW Mediterranean	2
Gyps fulvus	Griffon vulture		1
Lanius senator	Woodchat shrike	NW Mediterranean	2; 4
Limosa limosa	Black tailed godwit		6
Loxia curvirostra	Common crossbill	NW Mediterranean	2
Luscinia megarhynchos	Nightingale	NW Mediterranean	2; 4
Melanocorypha calandra	Calandra larks	Portugal	1; 5
Merops apiaster	European bee-eater	NW Mediterranean	2
Monticola solitarius	Blue rock thrush		4
Motacilla alba	White wagtail	Alps, Italy; NW Mediterranean	1;2;7
Muscicapa striata	Spotted flycatcher	NW Mediterranean	2
Oenanthe hispanica	Black-eared wheatear	NW Mediterranean	2
Oenanthe oenanthe	Northern wheater	NW Mediterranean	2
Oriolus oriolus	Eurasiam Golden oriole	NW Mediterranean	2; 4
Otis tarda	Great bustard	Portugal	1; 5
Passer domesticus	House sparrow		4
Perdix perdix hispaniensis (*)	Pyrenean Grey partidge		1
Perdrix perdrix	Grey partridge	North Savo; Picos de Europa	3
Petronia petronia	Rock sparrow	NW Mediterranean	2
Phoenicurus ochruros	Black redstart		4
Pyrrhocorax pyrrhocorax	Red-billed chough		1
Pyrrhula pyrrhula	Eurasian Bullfinch	NW Mediterranean	2
Saxicola rubetra	Whinchart	Alps, Italy	7; 1

Species	Vernacular	Location of the study	Ref.
Serinus serinus	European Serin	NW Mediterranean	1; 2; 4
Sylvia communis	Common whitethroat	NW Mediterranean	2
Sylvia conspicillata	Spectacled warbler	NW Mediterranean	2
Tetrao tetrix	Black grouse	Haut Jura	3
Tetrao urogallus	Capercaillie		8
Tetrax tetrax(**)	Little bustard	Portugal	1; 5
Turdus pilaris	Fieldfare		1
Turdus viscivorus	Mistle thrush	NW Mediterranean	2
Upupa epops	Hoopoe	NW Mediterranean	2; 4

C. Mammalia

Species	Vernacular	Location of the study	Ref.
Lepus corsicanus	Corsican Hare		1
Lynx pardinus	Iberian Lynx	Iberian peninsula	1
Oryctolagus cuniculus	European Rabbit		1

D. Reptilia

Species	Vernacular	Location of the study	Ref.
Acanthodactylus erythrurus (*)	Red-tailed Spiny-footed Lizard		1
Lacerta lepida (*)	Ocellated lizard		1
Natrix maura	Viperine water snake		1
Podarcis hispanica (*)	Iberian Wall lizard		1
Psammodromus algirus (*)	Large Psammodromus		1
Psammodromus hispanicus (*)	Spanish Psammodromus		1

E. Amphibia

Species	Vernacular	Location of the study	Ref.
Bombina variegata	Yellow-bellied toad		1
Discoglossus pictus	Mediterranean Painted Frog		1
Discoglossus sardus	Tyrrhenian Painted Frog		1

(*) Species that would benefit from early stages of re-vegetation post land abandonment but not from scrublands and woodlands.

(**) Species that are also considered as benefiting from land abandonment and rewilding in some locations.

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D. Appendix: Supplementary material to chapter 3.

Supplementary material for: Navarro L.M., Figueiredo J., Pereira H.M. (in prep). *A socio-ecological model of sedentarization: the case of brandas and inverneiras in Northern Portugal*.

List of parameters used in the socio-ecological model of sedentarization

Parameter	Description	Value
T_i	Total area in the *inverneiras* (ha)	100
T_b	Total area in the *brandas* (ha)	100
F_i	Forested area in the *inverneiras* (ha)	See Eq. (2) and (11)
F_b	Forested area in the *brandas* (ha)	See Eq. (3) and (12)
A_i	Agricultural area in the *inverneiras* (ha)	$T_i – F_i$
A_b	Agricultural area in the *brandas* (ha)	$T_b – F_b$
R	Total resident population (ind)	100
S	Sedentary population (ind)	See Eq. (10) and (13)
N	Nomad population (ind)	R - S
ε_i	Rate of forest growth in the *inverneiras*	low : 0.1 / high : 4
ε_b	Rate of forest growth in the *brandas*	low : 0.1 / high : 4
λ_i	Individual ability to cut the forest in the *inverneiras*	0.02
λ_{bN}	Nomads' individual ability to cut the forest in the *brandas*	0.02
λ_{bS}	Sedentaries' individual ability to cut the forest in the *brandas*	0.03
h_i	Utility of a unit of agricultural land in the *inverneiras*	2
h_b	Utility of a unit of agricultural land in the *brandas*	low : 1 / high : 3
σ	Utility from other sources than agriculture	1
c	Cost of nomadic lifestyle	0.9
ω	Probability to sedentarize (per ind, per unit of time)	1
μ	Individual treshold in the population	See Eq. (9)
δ	Social bonding	high : 0.1 / low : 0.5