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FOREST HABITAT MANAGEMENT AND CONSERVATION PRIORITIES: A MULTI-SCALE AND MULTI-TAXON APPROACH

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Summary

Habitat degradation, fragmentation and destruction are major causes of biodiversity loss. Management of natural and semi-natural habitats and control of human disturbance are fundamental to preserving their distinct character and biodiversity. Multiple levels must be considered when setting conservation management actions because species responses and ecological processes vary at different spatial scales. Legal instruments are now in place, with the European Union being among the pioneers, to protect and maintain habitats, and to implement management measures. Therefore, research efforts are needed to understand how to manage habitats in the current complex and constantly changing environmental and social context. For example, management of invasive alien species, which are among the most important threats to biodiversity, is a challenge nowadays. Furthermore, forest habitats are among the most important in terms of covered land and hosted species and, therefore, need particular attention. Indeed, several management approaches can be applied towards the achievement of biodiversity conservation objectives. However, the knowledge on the effects of different conservation management options on biodiversity is limited and must be further investigated.

The overall research follows a multi-disciplinary and integrated approach towards the conservation management of habitats particularly focusing on forest biodiversity. The thesis aims to (i) propose and test the application of integrated approaches in respect to conservation management of natural and semi-natural habitats focusing on forests, and (ii) to broaden the knowledge on the biodiversity effects of management abandonment. Six scientific papers, published and to be published, form the bulk of the thesis.

In the first paper a novel approach that aims to prioritize habitat conservation is proposed and tested in the Italian Alpine and Continental biogeographical regions. In the second paper a method is proposed and applied to assess the effects of human activities on habitats and species using as case study a forest road plan within a protected area. In the third paper a novel perspective on the potentiality of forest management to control invasive alien species is given. In the fourth paper a multi-scale landscape analysis was performed to identify habitat pattern changes due to different management regimes and to understand possible biodiversity implications. In the fifth paper a comparison between low intensity managed and abandoned forests was made to understand the effects on three beetle taxa. Finally, in the sixth paper the vegetation communities developing after management abandonment into novel forest habitats were investigated.

This thesis has highlighted that sound conservation management is fundamental to maintain the variety of habitats, both natural and semi-natural, occurring in Europe. On the one hand novel approaches, such as those presented in the thesis, are required to face the never-ending changes in the legal, economic, social and environmental conditions. On the other hand, deep knowledge on the effects of management and planning choices on habitats and species is essential for adapting to biodiversity's intrinsic variability and complexity in order to achieve conservation goals.

Riassunto

Titolo: Gestione degli habitat forestali e priorità di conservazione: un approccio multiscalare e multitassonomico

Le principali cause della perdita di biodiversità sono la degradazione, la frammentazione e la distruzione degli habitat naturali e semi-naturali. In tal senso, loro gestione è fondamentale per preservarne la diversità di caratteri distintivi, nonché per controllare gli impatti del disturbo antropico. Le azioni di conservazione della biodiversità devono essere individuate alle diverse e molteplici scale spaziali alle quali gli effetti della gestione sulle specie e sui processi ecologici si manifestano. La normativa europea, attraverso alcuni importanti strumenti giuridici per la protezione e il miglioramento degli habitat, richiede l'attuazione di specifiche misure di gestione. Ad oggi, le conoscenze degli effetti sulla biodiversità derivanti dalle attività gestionali risultano però limitate e lacunose. Pertanto è necessario che gli sforzi della ricerca siano focalizzati sulla gestione degli habitat, tenendo conto della dinamicità e complessità ambientale e sociale. Ad esempio, le specie esotiche invasive, una delle più importanti minacce alla biodiversità, rappresentano una odierna sfida a livello gestionale. Gli habitat forestali, tra gli altri, meritano una particolare attenzione, in quanto sono ampiamente diffusi e ospitano un elevato numero di specie.

La presente ricerca applica un approccio multi-disciplinare ed integrato alla gestione per la conservazione degli habitat, in particolare forestali. La tesi ha l'obiettivo di (i) proporre e testare l'applicazione di approcci integrati allo scopo non solo di identificare una gestione appropriata per la conservazione degli habitat naturali e semi-naturali, ma anche di (ii) conoscere in modo più approfondito gli effetti dell'abbandono delle attività antropiche sulla biodiversità. La tesi è composta da sei articoli scientifici pubblicati o in pubblicazione, dettagliati nel seguito.

Nel primo articolo viene presentato e discusso un nuovo approccio metodologico, applicato alle regioni biogeografiche italiane alpina e continentale, allo scopo di identificare gli habitat con maggiori esigenze gestionali finalizzate alla loro conservazione. Nel secondo articolo viene proposto un metodo utile a valutare gli effetti dell'attività antropica sugli habitat e sulle specie utilizzando come caso di studio un piano della viabilità forestale all'interno di un'area protetta. Il terzo articolo offre una visione sulle potenzialità che la gestione forestale può avere nel mitigare gli effetti delle specie esotiche invasive. Attraverso un analisi di paesaggio a più scale spaziali, nel quarto articolo si indentificano l'evoluzione degli habitat sottoposti a diverse intensità di gestione, evidenziandone le possibili implicazioni per la conservazione della biodiversità. Nel quinto articolo viene presentata una ricerca che analizza gli effetti dell'abbandono selvicolturale sulle comunità di tre gruppi tassonomici (carabidi, cerambicidi e scolitidi) e il loro habitat di specie. Infine, nel sesto articolo viene investigata la composizione delle comunità della flora vascolare dei boschi che si sono sviluppate a seguito dell'abbandono di aree urbane e peri-urbane.

Questa ricerca sottolinea l'importanza che la gestione ha nel mantenere la varietà degli habitat naturali che semi-naturali, in coerenza con gli obiettivi di conservazione europei.

Gli approcci innovativi, come quelli presentati nella tesi, sono indispensabili per adeguarsi ai cambiamenti delle condizioni giuridiche, economiche, sociali ed ambientali. Infine, una conoscenza approfondita degli effetti che la gestione e la pianificazione producono sugli habitat e sulle specie è essenziale per perseguire il mantenimento della complessità del paesaggio europeo.

1 Introduction

1.1 What do we mean by habitats?

Conserving biodiversity is a critical challenge that needs to be tackled. Biodiversity can be defined as “all terrestrial and freshwater organisms—including plants, animals, and microbes—at scales ranging from genetic diversity within populations, to species diversity, to community diversity across landscapes” (Sala *et al.*, 2000). Therefore, it represents a multi-scale concept (Lindenmayer *et al.*, 2006). Bearing in mind the complexity of biodiversity, measures of biodiversity are usually based on surrogates (Grantham *et al.*, 2010) such as groups of species and habitat types (Margules and Pressey, 2000).

The term habitat is recognized as one of the most important concepts and paradigms in ecology (Mitchell, 2005). Traditionally, habitat has been used to define an area with specific biotic and abiotic conditions where individuals of a species live (Whittaker *et al.*, 1973; Hall *et al.*, 1997; Kearney, 2006; Morrison *et al.*, 2006) (Fig. 1). Therefore, this term can be used to explain the association between elements of a landscape and species (Kearney, 2006).

Figure 1: Forest edge with abundant fleshy-fruit plants (here *Cornus sericea* – red osier dogwood) is habitat for the American black bear, *Ursus americanus* (Pallas) (British Columbia, Canada – picture: T. Campagnaro).

Furthermore, this term must not be confused with the concept of habitat type (Miller, 2000; Kearney, 2006; Miller and Hobbs, 2007). Habitat types refer to areas with similar vegetation associations (Daubenmire, 1968) (Fig. 2). This latter concept enables the mapping of extended areas by considering specific features that can be discriminated through the examination of aerial photos and other remote sensing results (Miller and Hobbs, 2007). Mapping habitat types provides spatial consistency which is useful for conservation planning (Margules and Pressey, 2000). However, several habitat classification schemes exist (e.g., IUCN habitat categories, Ramsar Wetland Type Classification System, EUNIS – European Nature Information System).

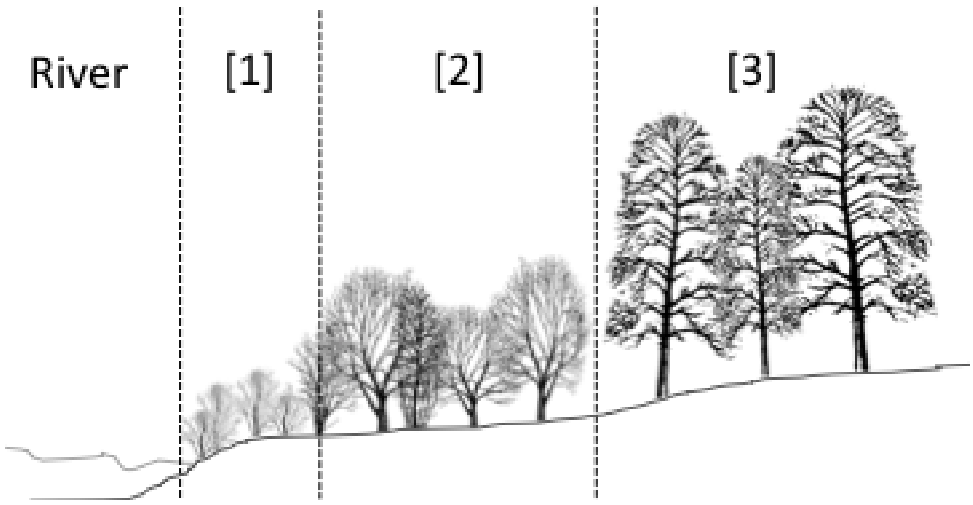

Figure 2: Representation of an example of habitat types according to the European Habitats Directive – [1] Alpine rivers and their ligneous vegetation with *Myricaria germanica*; [2] 3240 Alpine rivers and their ligneous vegetation with *Salix elaeagnos*; [3] 91E0* Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*, *Alnion incanae*, *Salicion albae*). These habitat types can be represented by several phytosociological units.

Habitats are a key component of biodiversity. Their extent and conditions are important indicators of the state of biodiversity (Butchart *et al.*, 2010). Globally recognized hotspots for conservation are areas where habitat loss and degradation is occurring (Myers *et al.*, 2000). Reduction of biodiversity is occurring at a multitude of spatial, temporal and biological levels (Tittensor *et al.*, 2014). Indeed, habitat degradation, fragmentation and destruction are a main cause of biodiversity loss (Brooks *et al.*, 2002; Butchart *et al.*, 2010). Not surprisingly, a common worldwide issue is the unprecedented habitat change in multiple directions (Suding and Hobbs, 2009). Furthermore, habitat loss and degradation will not see a substantial reduction in the next future (Tittensor *et al.*, 2014).

1.2 Habitat management

The establishment of protected areas is the most common and well-known practice to achieve biodiversity conservation goals (Pimm *et al.*, 2001; Ferraro and Pattanayak, 2006; Joppa *et al.*, 2008; Joppa and Pfaff, 2010) and they cover around 12 % of the earth's land surface (Joppa *et al.*, 2008). These areas are selected for their high conservation priority in which human activities and natural disturbances can be controlled (Gaspar *et al.*, 2011). Indeed, one best solution would be to establish a well-developed network of protected areas rather than managing protected areas as isolated habitats (Naughton-Treves *et al.*, 2005; Hole *et al.*, 2009). However, these areas require appropriate management measures in light of present and future pressures (Halpin, 1997; Thomas *et al.*, 2004).

In recent years, several international agreements aim to reduce changes in biodiversity (Sala *et al.*, 2000) highlighting an increase in the political effort to halt the loss of biodiversity (Geijzendorffer *et al.*, 2016). Habitats are a focal aspect for many international organizations and of several biodiversity-focused conventions, and international strategies.

For example, the International Union for Conservation of Nature (IUCN) considers habitat-related measurements to assess species' red listing (Brooks *et al.*, 2002).

One crucial aspect to preserve natural and semi-natural habitats within and outside protected areas is the implementation of appropriate conservation measures and management. Management effectiveness is important for achieving conservation goals within protected areas (Ervin, 2003; Hockings, 2003; Leverington *et al.*, 2010). Managing habitats means putting in place actions to influence habitat structure, processes and functions that will benefit specific species or assemblages of conservation interest (Ausden, 2007). Indeed, habitat management towards species conservation has reached significant results highlighting the importance of such management (Rands *et al.*, 2010). Moreover, habitat protection and management towards a single or a group of species will frequently advantage a whole set of species (Le Saout *et al.*, 2013).

Research highlights that deep ecological knowledge is required for the successful application of habitat management (New *et al.*, 1995). For example, deep knowledge of species ecology enables the use of habitat suitability models (Guisan and Zimmermann, 2000), which link species habitat preferences to spatial environmental information for management and impact assessment purposes (Guisan and Thuiller, 2005; Rondinini *et al.*, 2011). However, multiple levels must be considered when setting conservation management actions because species and ecological processes vary at different spatial scales (e.g., Doak *et al.*, 1992; Benton *et al.*, 2003; Opdam and Wascher, 2004; Pascual-Hortal and Saura, 2007).

Changes in management regimes and intensification of human activities can threaten species and habitat conservation at multiple scales (e.g., Chamberlain *et al.*, 1999; McCollin *et al.*, 2000; Zurlini *et al.*, 2006; McCollin and Geraghty, 2015). Indeed, avoiding human interference, as in wilderness preservation, is one way to maintain and enhance biodiversity (Sarkar, 1999; Höchtl *et al.*, 2005). Accordingly, rewilding has been highlighted as a possible approach towards biodiversity conservation (Navarro and Pereira, 2012; Ceauşu *et al.*, 2015; Pereira and Navarro, 2015; Svenning *et al.*, 2016). The term rewilding was initially linked to the restoration of viable population of large predators because of their regulatory role in the ecosystem (Soulé and Noss, 1998). Currently, it is intended as passive management (i.e. no management) enabling spontaneous succession with the ultimate goal of restoring natural processes and reducing human interference on landscapes (Sitzia *et al.*, 2010; Navarro and Pereira, 2012; Pereira and Navarro, 2015). However, focusing only on maintaining or establishing wilderness areas is argued not to be always the best solution for biodiversity conservation (Gómez-Pompa and Kaus, 1992; Sarkar, 1999; Svenning *et al.*, 2016).

Management is extremely important for semi-natural habitats. These are "nature-like" habitats deriving from various human actions (Ostermann, 1998; EEA, 2016). Indeed, current biodiversity patterns are shaped by management history (Cousins and Eriksson, 2002; Dupouey *et al.*, 2002; McCollin *et al.*, 2015). Therefore, it seems likely that these habitats will be maintained through the implementation of such management over time (Ostermann, 1998). Furthermore, heterogeneity at the landscape and local level were highlighted to be fundamental to maintain biodiversity (Webb, 1998; Cousins and Eriksson, 2002; Tscharntke *et al.*, 2005) as many species require a mosaic of habitats (Law and Dickman, 1998). This heterogeneity can be reached through appropriate management of these semi-natural habitats (Benton *et al.*, 2003; Tscharntke *et al.*, 2005). Indeed, a current priority for conservation is the management of different features of human-changed landscapes to maintain and restore habitats and their services (Chazdon, 2008; Gardner *et al.*, 2009; de Groot *et al.*, 2010)

1.2.1 The European Habitats Directive

The major legal instrument for protecting natural and semi-natural habitats is the European Habitats Directive (Tomaselli *et al.*, 2013). In 1992 the European Commission adopted the Habitats Directive (Directive 92/43/EEC). The sustainable protection of the European natural and semi-natural environment and wildlife is this Directive's final aim. This purpose is addressed by the establishment of Natura 2000 sites, that form the Natura 2000 network, and by the safeguard of specific species and habitats defined as of Community interest.

The Natura 2000 network is formed by Special Protection Areas (SPAs) – required under the Birds Directive (Directive 2009/147/EC is the codified version of Directive 79/409/EEC and its amendments) – and Sites of Community Importance (SCIs) approved and subsequently designated by the Member States as Special Areas of Conservation (SACs) – indicated under the Habitats Directive. SPAs are protected areas for rare and threatened bird species listed in Annex I of the Birds Directive and for migratory species; whereas, SCIs and SACs are sites aiming at conserving habitats and species of Annex I and Annex II of the Habitats Directive, respectively. Currently, this network includes 27,312 sites. The Habitats Directive also lists in its Annex IV species requiring strict protection in all EU countries within and outside Natura 2000 sites; whereas, in its Annex V reports species for which exploitation does not effect their conservation status. In total the Habitats Directive covers 233 habitats and approximately 1,250 flora and fauna species (EEA, 2015).

The Natura 2000 network is formed by set-a-side as well as private land on which sustainable management of natural resources should aim at conserving protected species and habitats (Annex I habitats and habitats of species) (Evans, 2012; Tsiafouli *et al.*, 2013). The Directive defines "habitat of a species" as "environment defined by specific abiotic and biotic factors, in which the species lives at any stage of its biological cycle", that recalls the original meaning of habitat.

As previously highlighted, one of the most innovative aspects of this directive is the identification and consequent required protection of habitats. In Europe, habitats are currently the backbone for biodiversity conservation because by maintaining or achieving a favourable condition for habitats we are able to safeguard protected species (Bunce *et al.*, 2013).

The Habitat Directive, under Article 1, defines natural habitats as "terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural" and highlighting that habitats of Community interest are those within the European Union territory which "(i) are in danger of disappearance in their natural range;

or (ii) have a small natural range following their regression or by reason of their intrinsically restricted area; or (iii) represent outstanding examples of typical characteristics of one or more of the nine following biogeographical regions: Alpine, Atlantic, Black Sea, Boreal, Continental, Macaronesian, Mediterranean, Pannonian and Steppic".

Annex I habitat classification can be defined as a hierarchical, consistent and exclusive system (Tomaselli et al., 2013). To identify these habitats the European Commission published interpretation manuals in several editions (EC, 2013) with their description. Nevertheless, different interpretations are given between regions (Evans, 2010) and single countries have published their own habitat manuals. The habitats are commonly defined by vegetation communities and usually following the phytosociological approach. For example, 26% or 53% of the habitats can be linked to one or more syntax by their name or description, respectively (Evans, 2010). This recognises the validity of using syntaxonomic categories as suitable synthetic ecological descriptors (Biondi *et al.*, 2012). Therefore, except for a reduced number of them that are landscape units, habitats are described through phytosociological alliances (Evans, 2006). The application of this approach within a regulation is one of the novel aspects of this Directive (Biondi *et al.*, 2012). Furthermore, this has had a positive effect of increasing the knowledge on several natural and semi-natural habitats as extensive surveys have been carried out around the European Union (Evans, 2006).

The Natura 2000 site designation process has mostly been completed and now attention is given towards setting conservation-sound management and protection measures (Ostermann, 1998; Evans, 2012). Indeed, research is needed to fill several of the knowledge gaps related to Natura 2000 (Blicharska *et al.*, 2016). Furthermore, additional effort is needed for the Alpine biogeographical region, forest habitats, and should encompass different spatial scales (Orlikowska *et al.*, 2016).

1.2.2 Invasive alien species: an important threat to habitats

Together with habitat loss and degradation, tackling invasive alien species is among the most important tasks that humanity is facing for protecting biodiversity (Diamond, 1989; Cardinale *et al.*, 2012). Indeed, habitat degradation can be caused or can cause the spread of invasive alien species (MacDougall and Turkington, 2005; Didham *et al.*, 2007). Invasive alien species are alien naturalized species able to spread over considerable area (Richardson *et al.*, 2000).

A large variety of different impacts on species and habitats have been detected (Vilà *et al.*, 2010; Vilà *et al.*, 2011; Pyšek *et al.*, 2012; Schirmel *et al.*, 2016). New combinations of species due to introduction and spread of alien species are forming "novel ecosystems" (Hobbs *et al.*, 2006; Kowarik, 2011). However, it is important to understand both negative and positive effects of invasive alien species for identifying best management solutions (Dickie *et al.*, 2014). Invasive alien species were highlighted to have several positive effects on ecosystem services in urban environments. (Kowarik, 2011; Sjöman *et al.*, 2016).

Worldwide, tree species are important invasive aliens (Richardson, 1998; Lamarque *et al.*, 2011; Richardson and Rejmánek, 2011). Alien tree species (example in Fig. 3) are considered an appropriate means for understanding general invasion processes (Petit *et al.*, 2004; Lamarque *et al.*, 2011). Furthermore, research on invasive alien tree species is important due to the relevant detrimental effects that they can have on the ecosystem properties and functions in natural and semi-natural woodlands (Richardson, 1998).

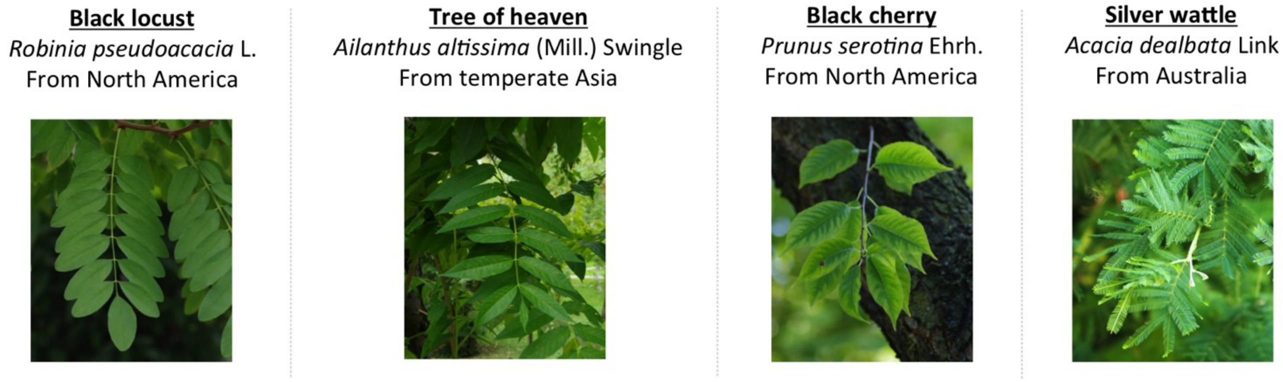

Figure 3: Four most threathening alien trees species to Europe (DAISIE, 2009) (Padova Botanical Garden, Italy – picture: T. Campagnaro).

Future scenarios highlight that the pressure from invasive alien species in Europe will continue to rise (Camenen *et al.*, 2016; Early *et al.*, 2016). Indeed, there is the need to link the knowledge on invasive alien species spread and impacts to management measures (Simberloff *et al.*, 2013). Recently, a better management of invasive alien species was requested in light of the new European Regulation 1143/2014 dealing with such issue (Pergl *et al.*, 2016).

1.3 Forest biodiversity and management

Forest habitats cover more than 30% of world's land (FAO, 2016). In Europe, according to the latest report available (Forest Europe, 2015), this proportion is 33% covering a total of 215 million ha. When considering only European Union countries this number grows to 38% for 161 million ha. This report highlights that within Europe these forest are divided into 87% semi-natural habitats, 3% undisturbed forests, and 9% plantations.

Forest habitats are the most important repository of biodiversity (FAO, 2016) hosting approximately 75% of terrestrial taxa (CPF, 2008). In Europe there are around 2100 tree species, of which one-quarter are under management for the provision of goods and ecosystem services (FAO, 2014). These habitats host a number of threatened species among several taxa (Forest Europe, 2015). Indeed, old-growth and ancient forests cover a small proportion of European forest, but are of extreme relevance for biodiversity (Rackham, 2008). Recent assessment on the conservation status of European protected habitats and species (EEA, 2015), highlights that 80% of reports on forest habitats indicate an unfavourable conservation status and that around 60% of assessments for forest species (excluding birds) show unfavourable conditions. For forest birds there is a more positive picture with 64% assessments reporting a secure population status.

Recently, in light of the aforementioned biodiversity importance of forests, there has been an increased awareness towards the conservation and sustainable management of forest resources (Winkel and Jump, 2014). Indeed, forests have a long history of use (e.g., Whitney and Davis, 1986; Denevan, 1992; Turner, 2001) and particularly in Europe (e.g., Rackham, 1976, 1980, 1986; Lindbladh and Bradshaw, 1998; Müllerová *et al.*, 2014; Szabó *et al.*, 2015). Clearly, distribution and tree composition of European forests has been influenced more by traditional management practices than by natural processes (Rackham, 1976, 1980; Peterken, 1981; Rackham, 2008; Munteanu *et al.*, 2016).

Differences and commonalities can be found in forest management approaches between continents but also neighbouring countries (Rackham, 1980; Peterken, 1981; O'Hara, 2001; Pommerening and Murphy, 2004; Puettmann and Ammer, 2007; Duncker *et al.*, 2012; Gustafsson *et al.*, 2012; Mori and Kitagawa, 2014) due to historical and ecological reasons. In recent years emphasis has been given to sustainable forest management (Peters and Schraml, 2014), an old, controversial concept (Wiersum, 1995) indicating the use of forests with the aim to maintain and enhance different forest functions, including the safeguarding of biodiversity (Rametsteiner and Simula, 2003). Moreover, nature conservation regulations affect forest planning in several countries (Cullotta *et al.*, 2015). Therefore, among other forests functions, biodiversity conservation is one main goal of forest management.

Managing for wood production, as well as for other services, does not necessarily result in negative biodiversity impacts. A recent study has shown a positive relationship between biodiversity and forest productivity suggesting the important value that biodiversity has in sustaining commercial forest productivity (Liang *et al.*, 2016). However, limited research was highlighted in many studies aiming to understand biodiversity relationships with management practices (Barbier *et al.*, 2008).

Forests are a complex system (Rowe and Scotter, 1973; Rackham, 1976; Bonan and Shugart, 1989; Kuuluvainen, 2009; Puettmann *et al.*, 2009) and management effects on biodiversity vary according to a multitude of factors such as disturbance severity, type of treatment, and its application (Roberts and Gilliam, 2003; Barbier *et al.*, 2008). Indeed, the modification and destruction of forest habitats due to human activities changes biodiversity at several spatial and biological scales (e.g., Grindal and Brigham, 1999; Chirici *et al.*, 2011; Newbold *et al.*, 2014). Mimicking the dynamics of natural disturbance has been highlighted as a solution for conserving biodiversity (Bengtsson *et al.*, 2000; Lindenmayer and McCarthy, 2002; Long, 2009). Many silvicultural practices have been proposed as following this perspective (e.g., Bergeron *et al.*, 1999; Gálhidy *et al.*, 2006). Furthermore, one option is to aim at old-growth forest characteristics (e.g., Keeton, 2006; Bauhus *et al.*, 2009; Barbati *et al.*, 2012). A review on impacts of European forest management on different taxa compared unmanaged and managed forests highlighting contrasting responses between different groups of species and indicating the need for more research on the effects of different silvicultural practices on several taxa (Paillet *et al.*, 2010). Moreover, a recent review has highlighted and described four possible forest management solutions for biodiversity conservation within Europe (Götmark, 2013). However, as stressed in this review, research is still needed to better understand the effects that different habitat management solutions have on biodiversity conservation (Lindenmayer *et al.*, 2000; Kuuluvainen, 2009; Götmark, 2013).

Another solution previously mentioned and in line with minimal intervention, is to passively rewild forests. This approach was argued to favour a variety of species occurring within forest habitats (Navarro and Pereira, 2012). Indeed, more empirical research is needed to attest its clear benefits for all biodiversity (Nogués-Bravo *et al.*, 2016), particularly for forests (Selva, 2016). Set-a-side forests left to natural processes will not enable to safeguard biodiversity if managed forests and other human uses outside protected areas are not part of a conservation strategy (Lindenmayer and Franklin, 2002; Branquart *et al.*, 2008).

Forest biodiversity and management outcomes can be monitored and assessed through specific indicators representing the most valuable features of these habitats for biodiversity (Lindenmayer, 1999; Noss, 1999). For example, among the most widely used are deadwood volume and type (Angelstam *et al.*, 2003; Schroeder *et al.*, 2006). Different silvicultural practices shape these biodiversity features (Brunet *et al.*, 2010). Indeed, for deadwood values, as well as for other indicators, research is needed for better understanding the implications for forest management aiming to preserve biodiversity (Bauhus *et al.*, 2009; Müller and Bütler, 2010).

Current day forest management requires novel and flexible measures to address old and new issues (Mori *et al.*, 2017). Indeed, current pressures and future threats to forests derive from the intensification of human activities and cessation of human activities that have influenced forests in the past (Rackham, 2008). The applied management practices need to be multi-scaled because different scales affect ecological processes, different species, and individuals of the same species (Lindenmayer, 2000; Lindenmayer *et al.*, 2006), and because planning and monitoring instruments act at different levels (Chirici *et al.*, 2011; Cullotta *et al.*, 2015).

1.3.1 Spontaneous forest expansion

Land use changes occur in a variety of ways with complex drivers and social, economic and ecological consequences (van Vliet *et al.*, 2016). Even tough deforestation continues to be a worrying problem (Gibson *et al.*, 2011); in many areas of the world we are now facing forest transitions (Rudel *et al.*, 2005; Rudel *et al.*, 2009). Indeed, the threat of losing old-growth and ancient forests is still high also in Europe (e.g., Rackham, 2008; Chylarecki and Selva, 2016; Kindlmann and Krenova, 2016). However, forest transition indicates a change from net loss to gain in forest area deriving from multiple land changes shifts at the national level (Meyfroidt and Lambin, 2011). Reforestation usually occurs through the establishment of new plantations or through spontaneous forest succession.

Land abandonment is a worldwide phenomenon (MacDonald *et al.*, 2000; Hobbs and Cramer, 2007; Rey Benayas *et al.*, 2007; Haddaway *et al.*, 2014). After abandonment, natural succession occurs and forest spontaneously covers land over time (Sitzia *et al.*, 2010). Management abandonment is occurring throughout most of Europe and this is forecasted to continue in the future (Lasanta *et al.*, 2017). Indeed, these processes can have different and contrasting effects on biodiversity (e.g., Barlow *et al.*, 2007; Bowen *et al.*, 2007; Dent and Joseph Wright, 2009).

In Europe, abandonment coupled with forest expansion has been stressed to cause negative impacts on biodiversity (Sala *et al.*, 2000; Sitzia *et al.*, 2010; Queiroz *et al.*, 2014). For this reason, it has been suggested to apply integrative (Rey Benayas and Bullock, 2012) and active management to these lands in order to restore pre-abandonment biodiversity conditions (Ascoli *et al.*, 2013; Lasanta *et al.*, 2015). Nevertheless, studying spontaneous reforestation is fundamental as forest distribution could impact habitat functions and processes at different spatial scales (Rudel *et al.*, 2005) and further research efforts are needed for a better understanding of such effects on biodiversity (Meyfroidt and Lambin, 2011) and on landscape patterns (Sitzia *et al.*, 2010).

Abandonment and forest expansion processes can potentially benefit the spread of invasive alien species. Many alien tree species have been found invading abandoned agricultural and urban land patches (Sitzia *et al.*, 2012; Trentanovi *et al.*, 2013). The high habitat suitability for major invasive alien tree species in Europe under the current and future climate (Camenen *et al.*, 2016) suggests that if abandonment processes continue forests formed by these species will expand.

1.4 Research objectives

The general aims of this thesis are:

i. To propose and test the application of integrated approaches in respect to conservation management of natural and semi-natural habitats focusing on forests
ii. To broaden the knowledge on biodiversity effects of management abandonment with a particular focus on forests

Specifically, the research has the following objectives:

i. a) To propose a method for prioritising conservation management of natural and semi-natural habitats
   b) To propose a method for assessing human activities' impact on habitats
   c) To understand the role that forest management can play in controlling invasive alien species
ii. d) To identify habitat pattern changes at different spatial scales due to management abandonment
   e) To quantify changes in composition and diversity of different species groups due to forest management abandonment
   f) To assess the effects of forest expansion due to neglected management on plant communities

1.5 Research framework

This research follows a multi-disciplinary approach in the study of the conservation management of semi-natural habitats and falls within a wide spectra of scientific fields: biodiversity conservation, environmental management, ecology, landscape ecology, and planning. Specific attention is given to forest habitats because they host a great proportion of world's biodiversity and, therefore, require research for their management. The thesis investigates the application of different approaches in habitat management with a link to the most important EU policies and regulations regarding biodiversity conservation and management and focuses on effects of management at multiple scales and on several taxa (Fig. 4).

The thesis is base on six main chapters, each representing different research papers. These are research papers that have been published or are in preparation for submission to scientific journals. Overall the set of papers cover the general aims of the thesis. Each chapter focuses on specific aspects representing the specific objectives.

Paper I – Chapter 2

"Identifying habitat conservation priorities under the Habitats Directive: application to two Italian biogeographical regions"

This chapter focuses on the first aim (objective *a*) and presents the proposal of a new integrated method that enables to identify conservation management priorities for natural and semi-natural habitats under the European Habitats Directive.

Paper II – Chapter 3

"Ecological risk and accessibility analysis to assess the impact of roads under Habitats Directive"

This chapter focuses on the first aim (objective *b*) and tests the application of a method to assess impacts of human activities (i.e. forest road plans) on natural and semi-natural habitats (i.e., forests and their biodiversity) under the requirements of Article 6(4) of the Habitats Directive.

Paper III – Chapter 4

"Using forest management to control invasive alien species: helping implement the new European regulation on invasive alien species"

This chapter focuses on the first aim (objective *c*) and gives a perspective on the important role that forest management can play to combat invasive alien tree species in light of the European Regulation on invasive alien species.

Paper IV – Chapter 5

"Multi-scale analysis of alpine landscapes with different intensities of abandonment reveals similar spatial pattern changes: implications for habitat conservation"

This chapter focuses on the second aim (objective *d*) and investigates the effect of different management regimes (low-intensity vs. abandoned) at the landscape level on habitats over time and at different spatial scales.

Paper V – Chapter 6

“Wildlife conservation through forestry abandonment: responses of beetle communities to habitat change in the Eastern Alps”

This chapter focuses on the second aim (objective *e*) and investigates the responses of three beetle taxa to different forest management regimes (low-intensity vs. abandoned). A focus is given to species richness, abundance and composition. Responses to specific habitat features were also investigated because management can shape these features.

Paper VI – Chapter 7

“Novel woodland patches in a small historical Mediterranean city: Padova, Northern Italy”

This chapter focuses on the second aim (objective *f*) and studies plant diversity within novel forests growing in abandoned areas within an urban setting. It gives an overview of the plant communities characterised by alien species but also typical native communities. Furthermore, effects patch size, stand, and urbanization on these plant communities were investigated.

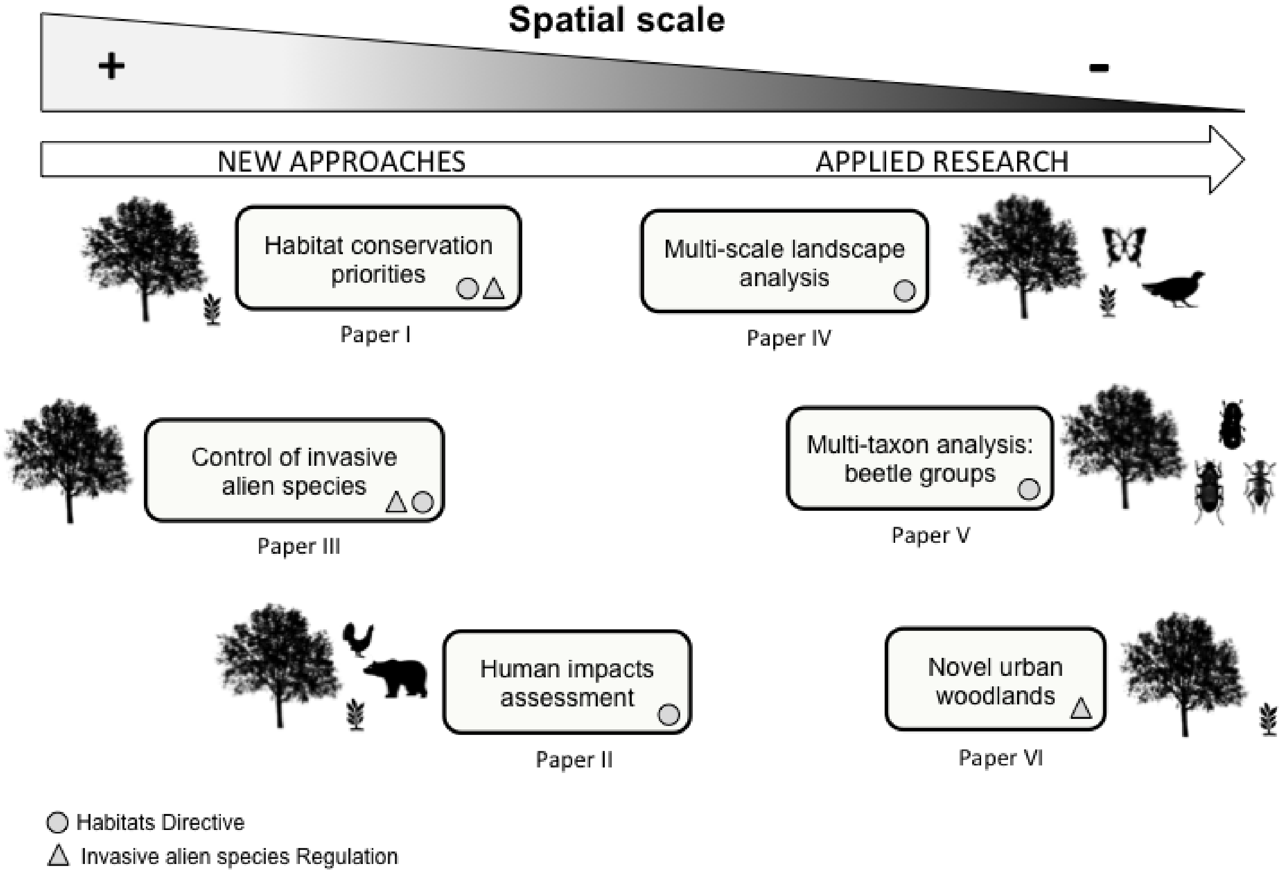

Figure 4: Schematic research framework. Paper titles have been reduced for space necessity but they summarise the main topic.

The thesis deals both with methodological and applied research to investigate habitat conservation management. To contain both these aspects the research was divided into different topics that fall in these two broad categories represented by the two general objectives. The first group of papers (paper I, II, and III) represent proposals of novel methods and approaches that find application at different scales according to the current European legal framework concerning biodiversity. The problem of habitat conservation management is approached in the context of the most important pieces of legislation at the European Union level: the Habitats Directive (92/43/EEC) (hereafter Directive) and the invasive alien species (IAS) Regulation (No. 1143/2014) (hereafter Regulation). However, particularly for paper II, field data is used to test the methodology.

The second group of papers represent empirical and applied research (IV, V, and VI) focusing on the understanding of how management can have an influence on habitats. These three papers focus on the effects of management abandonment with the spontaneous development of forests. These papers are also linked to the Directive and Regulation. In paper IV protected habitats and species under the Directive are considered while in paper V the focus is on a single protected forest habitat. In paper VI the presence of several invasive alien species in the forest communities indicates an important context for the application of the Regulation.

The thesis focuses on different spatial scales and levels. The single chapters, in certain cases, overlap in terms of spatial extent considered because different scales must be considered when dealing with regulatory requirements and ecological processes. Furthermore, one paper (IV) specifically analyses the effects of changing spatial extent on habitat landscape indexes. From the widest scale to the narrower we can distinguish the papers as follows: in paper III the entire European Union territory is considered as representing the regulation context, in paper I the proposed approach is tested at the national biogeograhical scale because of the requirements of the Directive, in paper II the assessment method is applied to the planning scale, in paper IV two watersheds are investigated to understand habitat patterns at different extents, in paper V two forests of the abovementioned watershed are analysed, and in paper VI small woodlands are investigated within the boundaries of a historical city.

The thesis takes into account several taxa within the different papers. In all papers forest habitats are taken into consideration. In paper I and IV also other habitats are particularly considered. Plants communities and species are an important part of the analysis carried out in paper I and VI. Instead, in paper II animal (mammals and birds) and plant species are important features in the assessment of human impacts. In paper IV implications for the conservation of protected species (butterflies and birds) found in the two watersheds is discussed. In paper V a specific focus is given to ground, longhorn and bark beetles.

2. Paper I: Identifying habitat conservation priorities under the Habitats Directive: application to two Italian biogeographical regions[1]

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Abstract

Due to the ongoing biodiversity crisis, efforts are needed to ensure a good conservation of habitats. In Europe, conservation focus is now shifting from identifying areas and biodiversity features to be protected to the management needs of these areas and habitats. This issue is particularly important for the conservation of natural and semi-natural habitats under the Habitats Directive framework. Here we proposed an approach to determine conservation management priorities for habitats based on readily available data. We tested the method by focusing on the habitats occurring within the Italian Alpine and Continental biogeographical regions. A set of four simple criteria, i.e. conservation condition, biodiversity value, affecting pressures, and cover relevance of habitats, with related representative parameters was used to rank habitats and identify related management requirements. After ranking habitats based on the sum of scores given to all criteria, habitats conservation was prioritized. The affecting pressures are analyzed through cluster analysis to better convey information on management needs of groups of habitats. These pressures were then used to suggest management measures for habitats of conservation priority. Forests, bogs and fens, and dry grasslands are conservation priorities for the Alpine region; whereas, a wider variety of habitats were highlighted for the Continental region. Important conservation measures were identified for these two biogeographical regions of Italy; for example, appropriate measures must tackle the high pressure posed by roads an motorways. The approach enabled to transparently outline possible conservation measures for prioritized habitats with the potential to help achieving biodiversity targets.

Keywords: Biodiversity conservation, Conservation status, Conservation management, Prioritization, Natura 2000, biodiversity indicator

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[1] Edited version of the paper under review in *Environmental Management*

2.1 Introduction

Currently, the lack of funding together with the ongoing biodiversity crisis is calling for the prioritization of conservation management efforts in order to ensure natural and semi-natural habitats for future generations. Worldwide an increasing number of political efforts are made to stop the loss of biodiversity (Geijzendorffer et al. 2015). Indeed, European society is strongly concerned on the loss of biological diversity; therefore, stopping this phenomenon is viewed as a critical challenge in Europe (Hochkirch et al. 2013).

In 1992 the European Commission (EC) adopted the Habitats Directive (Directive 92/43/EEC) that is the most important legislative accomplishment relevant to biodiversity conservation (Maiorano et al. 2006; Tomaselli et al. 2013). The Directive aims at a sustainable protection of the European natural and semi-natural habitats, flora and wildlife. This Directive foresees the establishment of the Natura 2000 network that, together with the legal protection of habitats and species, is one of the largest conservation areas worldwide (Sundseth and Creed 2008; EEA 2012). Habitats and species of Community interest, for which measures are needed towards their protection and restoration, are listed in Annex I and II, IV and V (see the Habitats Directive for specific definitions). Article 17 of this Directive requires Members States to periodically report (i.e. every six years) on the conservation status of habitats and species and on the conservation measures undertaken on their territory. This data must be reported separately for 9 terrestrial and 5 marine biogeographical regions.

According to the last European Union (EU) composite report based on these national reports, 30% of habitat and 18% species assessments, are in a bad conservation status at the EU scale (EEA 2015). Furthermore, target 1 of the EU 2020 Biodiversity Strategy requires that 100% or more of habitat and 50% or more of species assessments be favourable or improving compared to the composite report for period 2001-2006 (EC 2011). In this respect, progress is still much needed both for species and habitats (EEA 2015).

Conservation problems and negative trends should be addressed through strategic planning, by the identification of appropriate management measures, and sound prioritization of the latter. Many studies at the EU level have underlined priorities in terms of taxa and areas to be further protected after investigating Natura 2000 gaps, effectiveness, and representativeness at different spatial scales (Maiorano et al. 2006; Maiorano et al. 2007; Jantke et al. 2011; D'Amen et al. 2013; Popescu et al. 2013; Votsi et al. 2016). Furthermore, many different approaches have been proposed to prioritize species conservation (e.g., Schnittler and Günther 1999; Bani et al. 2006; Martín et al. 2010; Gauthier et al. 2013). However, relatively little focus has been given to habitats and to the conservation measures they require to achieve or maintain a good conservation status.

Habitats are usually ranked by considering the biodiversity value of their plant communities (e.g., Bragazza 2009; Angiolini et al. 2016) and by considering national responsibility for each habitat (e.g., Schmeller et al. 2012; Schmeller et al. 2014). For example, Bacchetta et al. (2012) ranked habitats based on their endemic plant richness and on their related priority index value. Usually attention has been given to assessing habitat conservation priorities by focusing at the Natura 2000 site and network level (e.g., Graziano et al. 2009; Velázquez et al. 2010; Mikkonen and Moilanen 2013). These approaches focus on prioritizing conservation among habitats, but lack of direct link to pressures influencing the conservation status of habitats and on possible conservation measures (but see risk assessment approaches; Foresta et al. 2016; Sitzia et al. 2016a). Indeed, a large variety of human activities occur in Natura 2000 sites and may impact habitats (Tsiafouli et al. 2013). Furthermore, novel approaches should consider data availability together with the assessment requirements of Habitats Directive notwithstanding the spatial scale appropriate for the regulatory framework. Recently, new approaches to achieve quick conservation benefits at the biogeographical level were proposed ("low-hanging fruit" approach; Richard et al. 2016). Under the Habitats Directive several spatial scales are important for administrative and management reasons: European Union, biogeographical region, Member State, Natura 2000 site and their possible combinations (e.g., biogeographical region of a Member State). However, identifying priorities across administrative scales is difficult (Schatz et al. 2014). In the context of Habitats Directive, future actions should prioritize conservation measures to attain and maintain favourable conservation status of habitats and species (Maiorano et al. 2015). Indeed, there is a demand for scientific support on identifying conservation priorities and feasible management options (Pullin et al. 2009; Popescu et al. 2014; Louette et al. 2015).

Here, our goal was to develop a simple and objective method to prioritize conservation of natural and semi-natural habitats and help suggesting appropriate conservation management measures. The method was based on the combined evaluation of habitats in terms of conservation condition, biodiversity value, affecting pressures and cover relevance by capitalizing on data available from assessments under Article 17 of the Habitats Directive. To test our ranking approach we identified conservation management priorities for habitats of the Alpine and Continental biogeographical regions of Italy.

2.2 Methods

2.2.1 Study area

The method was tested for the habitats occurring within the Alpine and Continental biogeographical region of Italy. The Alpine biogeographical region covers the Italian Alpine range and two relatively small areas in central Italy within the Appenines (Maiella massif and Gran Sasso mountain). In Italy this biogeographical region covers around 51000 km^2. The Alps are a complex mountainous system interspersed by long valleys and Alpine rivers. The last national report (Genovesi et al. 2014) assessed 76 habitats, 47 plants and 116 animal species listed in Annex I, II, IV and V. The most represented habitat category is forests (26) followed by grasslands (13), freshwater habitats (12) and rocks and screes (9).

The Continental biogeographical region covers the Po plain and parts of the Adriatic coast. This biogeographical region covers around 88000 km^2. The last national report assessed 83 habitats, 38 plants and 124 animal species listed in Annex I, II, IV and V (Genovesi et al. 2014). The most represented habitat category is forests (21) followed by grasslands (13), water habitat (12) and dunes (9).

2.2.2 Methodological framework

The method applied to identify habitat conservation priorities is based on a set of 4 criteria. The value of each criteria was standardized; therefore they range from 0 to 1. Nevertheless, weights can be given to these criteria to underline their different importance. The four criteria are: (i) conservation condition, (ii) biodiversity value, (iii) affecting pressures, and (iv) cover relevance (Fig. 1). All these criteria are related to specific parameters derived from the official data reported to the European Commission by Italy (data is presented and summarized in Genovesi et al. 2014). Member States reports under Article 17 for all species and habitats can be found at http://bd.eionet.europa.eu/activities/Reporting/Article_17. This data is the most comprehensive currently available for habitats of Community interest.

Each parameter is valued through a scoring approach (Table 1). The scoring approach is adopted in other methods for assessing conservation priorities (e.g. Schmeller et al. 2008a). The final score of the criteria derives from the scalar value of the summed values assigned to the parameters. To enable comparisons, the value of each criteria is scaled by the highest value (Cain and Harrison [1958] as cited in Legendre and Legendre 1998). These criteria are then used to detect priorities by identifying the habitats “most in need”, i.e. those habitats for which prompt action through management is highly important.

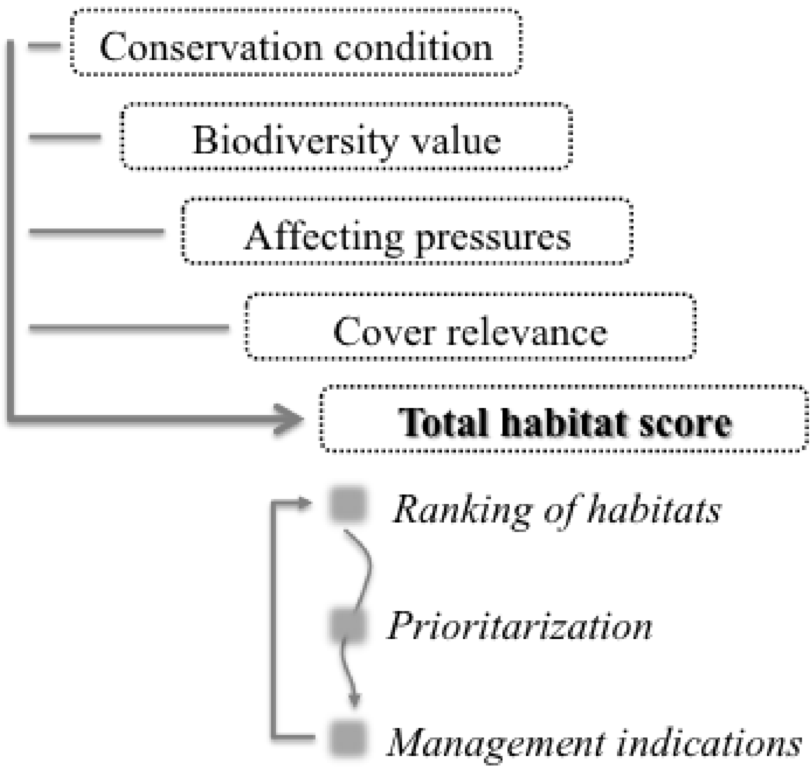

Figure 1: Steps of the conservation management prioritization method

2.2.2.1 Conservation condition

Conservation status has been already used to prioritize species and habitat conservation (e.g., Gauthier et al. 2013; Mikkonen and Moilanen 2013). EU Member States are required to report on conservation status of habitats listed in Annex I. The overall conservation status of one habitat in a biogeographical region derives from separate evaluations of 4 parameters: range, area, structure and functions, and future prospects (Evans and Arvela 2011). The assessment of each parameter is one of four classes (FV= Favorable, U1= Unfavorable/Inadequate, U2=Unfavorable/Bad, XX=Unknown). The overall value for a single habitat usually is derived from the lowest value among the different parameters. However, while it is clear that this approach aims at stimulating an improvement towards a good conservation status, it is not completely suitable for assessing priorities. For example, the overall conservation status of a habitat with only one parameter in bad condition is equal to that of a habitat with all parameters in a bad condition. Nevertheless, from a management perspective there is an important difference between these two examples. In the former case to improve the overall value only the condition of the one parameter must change, whereas, in the latter all four must improve. Therefore, theoretically, efforts should be greater in the latter case rather than in the former one. Furthermore, this difference indicates a real difference in the overall condition.

Here we assign to each parameters a score (similarly to Richard et al. 2016) between 0-1 with higher values indicating worse conditions. To distinguish between these possible cases, the conservation criterion is defined by the sum of the values of the single parameter:

$$HCC_i = R_i + A_i + S_i + F_i \tag{1}$$

where HCC_i is the conservation condition of habitat *i*, *R* is the score assigned to the range of habitat *i*, *A* is the score assigned to the area covered by habitat *i* within its range and with reference to a favorable reference area (see Evans and Arvela 2011), *S* is the score assigned to the structure and function of habitat *i*, and *F* is the score assigned to the future prospects of habitat *i* (see Table 1 for specific scores).

2.2.2.2 Biodiversity value

Habitats are indicators of biodiversity (Bunce et al. 2013). Therefore, when prioritizing conservation efforts it is important to underline the habitats significance in terms of biodiversity conservation value. Habitats (and species) listed in the Habitats Directive are distinguished between priority (identified with an asterisk) and non priority. Priority habitats can be considered to have an added biodiversity value as these, according to Article 1, are “in danger of disappearance” and are considered being of European “particular responsibility”. Furthermore, this priority by definition indicates the most vulnerable habitats (Gauthier et al. 2013); even though, at regional and national level not always these habitats are to be considered as such (e.g., bushes with *Pinus mugo* and *Rhododendron hirsutum* (*Mugo-Rhododendretum hirsuti*)) because of their wide distribution. To consider this added value, our method assigns a score (1) to priority habitats.

Furthermore, habitats can be differentiated based on their biodiversity components as many taxa are associated to them (Bunce et al. 2013). Vegetation classification methods were widely used to define habitats of European interest (Evans 2006) and lists of characteristic plant species (not always with a phytosological meaning) is available at the European (EC 2013) and country level (e.g. Biondi et al. 2009). Furthermore, typical species are considered within the reporting under Article 17. The consideration of plant species of particular concern within a habitat also enables to promote species conservation. Several prioritization approaches consider the presence of threatened species within habitats as a proxy of habitat conservation value (e.g., Bragazza 2009; Mikkonen and Moilanen 2013; Berg et al. 2014). Therefore, one can identify the species among those underlined of particular concern (e.g., red listed and policy species) usually found within a habitat. Here we screened the list of typical species reported for the single biogeographical regions of Italy to identify species listed in Annex II, IV and V of Habitats Directive and in the national Red List (Rossi et al. 2013). Based on these lists, scores were assigned to species if they are reported in the Habitats Directive Annexes and red list categories, and whether they are endemic (Berg et al. 2014).

The biodiversity value derives from

$$HCV_i = P_i + A_i + Rl_i + E_i \quad (2)$$

where the biodiversity value of habitat *i* is the score assigned whether it is a priority habitat (P) and the sum of the scores (Table 1) for species in the Habitats Directive Annexes (HD), in the national red list (Rl), and whether there are endemic species (E). The scores assume higher values for species of European interest and lower for species of national interest (see Table 1 for specific scores).

2.2.2.3 Affecting pressures

Here affecting pressures are meant as external factors acting with a detrimental effect on the habitat. To identify this criterion, both current acting pressures and future acting pressures are considered. The identification of the disturbances affecting the habitats is essential for their management. A significant problem in the conservation of habitats is the attenuation of the most important impacting factors (Fenu et al. 2015). Impacting factors on habitats were used by Graziano et al. (2009) to detected and prioritize conservation strategies at the Natura 2000 site level.

We considered the list of pressures and threats that can affect habitats and that is used for reporting under Article 17. This list derives from the combination of different lists (e.g., Salafsky et al. 2008) reporting specific actions that can be detrimental. While pressures refer to forces currently impacting on the habitat, threats indicate forces that will impact the habitat in the future (Evans and Arvela 2011). Indeed, considering both present and future impacting actions is crucial to identify priorities.

At each identified pressure, whether current or future, the assessment assigns three possible categorical values (high, medium, low) based on their relative importance. These values take into consideration intensity and area of influence. The value of the affecting pressures to a habitat was derived from the sum of the value assigned to the impact factors identified.

The affecting pressures derives from

$$IF_i = \sum H_p + M_p + L_p + H_t + M_t + L_t \quad (3)$$

where the impacting pressures of habitat *i* is the score deriving from the sum of high (*H*), medium (*M*) and low (*L*) scores for pressures ($_p$) and threats ($_t$) (see Table 1 for specific scores).

2.2.2.4 Cover relevance

Many approaches to prioritize species conservation contain a criteria based on spatial parameters (e.g., Gauthier et al. 2010; Martín et al. 2010; Gauthier et al. 2013; Benavent-González et al. 2014; Schatz et al. 2014). For example, Louette et al. (2011) set priorities on regional conservation objectives considering the relative contribution of Flanders to the area covered by a habitat within the European Union.

The cover relevance criterion (CR) is related to the geographical distribution of the different habitats; it derives from the area covered by the habitat. This criterion is similar to that of the regional (Benavent-González et al. 2014) and national (Schmeller et al. 2014) responsibility approaches. Cover relevance indicates how much area is covered by the habitat in that country's biogeographial region compared to the total area reported for the same biogeographical region.

The cover relevance derives from

$$SR_i = \frac{A_{Ci}}{A_{Ti}} \quad (4)$$

where the cover relevance of habitat *i* is the score deriving from the proportion of the area cover within the country of analysis (A_C) and the cover at the EU level (A_T) for the same biogeographical region.

Table 1: Selected criteria, parameters, values and related scores

Criteria	Parameters	Values	Scores
Conservation condition	- Range (R) - Area (A) - Structure and functions (S) - Future prospects (F)	Favorable	0
		Unknown	0.33
		Inadequate	0.66
		Bad	1
Biodiversity value	Priority (P)	No	0
		Yes	1
	Species of the Habitats Directive Annexes (HD)	Annex V	0.25
		Annex IV	0.5
		Annex II	0.75
		Priority Annex II	1
	Red list species (Rl)	Least Concern	0.2
		Near Threatened	0.4
		Vulnerable	0.6
		Endangered	0.8
		Critically Endangered	1
	Endemic species (E)	No	0
		Yes	0.5
Affecting pressures	- Pressures (p) - Threats (t)	Low	0.33
		Medium	0.66
		High	1
Cover relevance	Cover relevance (SR)	Area proportion	0-1

2.2.2.5 Total habitat score

The values assigned to the four criteria are summed for each habitat. These values were previously scaled considering the maximum value recorded to have a final possible range of values between 0 and 4. Then, habitats were ranked based on this value.

2.2.2.6 Habitat ranking and conservation priorities

Habitats were then ranked based on the total habitat score. To identify management actions for habitats of conservation priority, the decision of selecting the first ranking habitats (25% of the total) was made. This percentage is subjective and can be changed based on political, social and economic requirements. For these habitats an analysis of the main affecting pressures was carried out. This together with a backwards analysis of the conservation condition made it possible to underline which conservation measures should be favored in order to have a beneficial effect on the habitat.

However, it can be possible to have beneficial effects on several habitats by applying single conservation measures. Therefore, a cluster analysis was performed to understand whether there were groups of habitats subjected to similar pressures. This was applied to enable formulate detailed conservation measures (Zhang et al. 2014). We used ordinal data corresponding to the different degrees of pressure and threats (i.e., raw values before scaling). These values were then summed to have unique value of affecting pressures. All analyses were performed using R statistical programme (R Development Core Team 2015). As suggested for analyzing ordinal data (Fabbris 1997), we used the city-block distances that are the sum of absolute differences by applying the “Manhattan” function. Data was analyzed with an agglomerative hierarchical clustering by using Ward’s clustering method. To understand which affecting pressures were those to be prioritized, we identified the affecting pressures shared by groups of habitats. First, we identified the most common affecting pressures (more than 50% of habitats) and, then, we identified groups from the cluster analysis that adequately represented the habitats. To investigate which affecting pressures defined these groups, we applied the Indicator Species Analysis (ISA) (Dufrêne and Legendre 1997). We used the “Multipatt” function of the “indicspecies” package in R software (De Cáceres et al. 2010) as the indicator value method (IndVal). The Monte Carlo test with 999 randomizations was used to verify statistical significance. Group combinations were considered.

2.3 Results

The method allowed habitats in the Alpine and Continental biogeographical regions of Italy to be ranked. In table 2 and 3 the habitats ranking for the Alpine and Continental biogeographical regions are reported. In the Alpine biogeographical regions the higher-ranking habitats (first 19) were represented by forest (10), bog and fen (5), and grassland (4) habitats. Whereas, in the Continental biogeographical region, the higher-ranking habitats (first 21) were represented by a wider variety of habitats: dunes (5), coastal (5), forest (4), grassland (3), freshwater (2), and fen (2) habitats.

Table 2: Habitats of the Italian Alpine biogeographical region ranked with the proposed method (* highlight priority habitats *sensu* Habitats Directive)

Rank	Habitat code	Habitat name	Priority
1	6240	*Sub-Pannonic steppic grasslands	2.70
2	7230	Alkaline fens	2.47
3	91H0	*Pannonian woods with *Quercus pubescens*	2.41
4	62A0	Eastern sub-Mediterranean dry grasslands (*Scorzoneratalia villosae*)	2.39

Rank	Habitat code	Habitat name	Priority
5	91F0	Riparian mixed forests of *Quercus robur*, *Ulmus laevis* and *Ulmus minor*, *Fraxinus excelsior* or *Fraxinus angustifolia*, along the great rivers (*Ulmenion minoris*)	2.32
6	91E0	*Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion, Alnion incanae, Salicion albae*)	2.32
7	6210	Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (* important orchid sites)	2.31
8	7110	*Active raised bogs	2.25
9	91L0	Illyrian oak-hornbeam forests (*Erythronio-Carpinion*)	2.20
10	7240	*Alpine pioneer formations of *Caricion bicoloris-atrofuscae*	2.11
11	92A0	*Salix alba* and *Populus alba* galleries	2.09
12	9160	Sub-Atlantic and medio-European oak or oak-hornbeam forests of the *Carpinion betuli*	2.03
13	7220	*Petrifying springs with tufa formation (*Cratoneurion*)	2.02
14	7210	*Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae*	2.02
15	9260	*Castanea sativa* woods	2.01
16	9510	*Southern Apennine *Abies alba* forests	1.97
17	91AA	*Eastern white oak woods	1.97
18	6510	Lowland hay meadows (*Alopecurus pratensis, Sanguisorba officinalis*)	1.87
19	91D0	*Bog woodland	1.83
20	3170	*Mediterranean temporary ponds	1.78
21	6410	*Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (*Molinion caeruleae*)	1.78
22	7140	Transition mires and quaking bogs	1.72
23	6520	Mountain hay meadows	1.72
24	9180	**Tilio-Acerion* forests of slopes, screes and ravines	1.70
25	3260	Water courses of plain to montane levels with the *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation	1.69
26	6110	*Rupicolous calcareous or basophilic grasslands of the *Alysso-Sedion albi*	1.66
27	3230	Alpine rivers and their ligneous vegetation with *Myricaria germanica*	1.66
28	6230	*Species-rich *Nardus* grasslands, on siliceous substrates in mountain areas (and submountain areas, in Continental Europe)	1.66
29	9220	*Apennine beech forests with *Abies alba* and beech forests with *Abies nebrodensis*	1.64
30	8340	Permanent glaciers	1.62
31	6420	Mediterranean tall humid grasslands of the *Molinio-Holoschoenion*	1.62
32	9530	*(Sub-) Mediterranean pine forests with endemic black pines	1.61
33	9210	*Apennine beech forests with *Taxus* and *Ilex*	1.60
34	7150	Depressions on peat substrates of the *Rhynchosporion*	1.58
35	9420	Alpine *Larix decidua* and/or *Pinus cembra* forests	1.56

Rank	Habitat code	Habitat name	Priority
36	8230	Siliceous rock with pioneer vegetation of the *Sedo-Scleranthion* or of the *Sedo albi-Veronicion dillenii*	1.46
37	3280	Constantly flowing Mediterranean rivers with *Paspalo-Agrostidion* species and hanging curtains of *Salix* and *Populus alba*	1.40
38	9430	Subalpine and montane *Pinus uncinata* forests (* if on gypsum or limestone)	1.38
39	4070	*Bushes with *Pinus mugo* and *Rhododendron hirsutum* (*Mugo-Rhododendretum hirsuti*)	1.35
40	4080	Sub-Arctic *Salix* spp. scrub	1.32
41	6220	*Pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea*	1.25
42	3270	Rivers with muddy banks with *Chenopodion rubri* p.p. and *Bidention* p.p. vegetation	1.23
43	3110	Oligotrophic waters containing very few minerals of sandy plains (*Littorelletalia uniflorae*)	1.22
44	6430	Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels	1.19
45	9410	Acidophilous *Picea* forests of the montane to alpine levels (*Vaccinio-Piceetea*)	1.18
46	91K0	Illyrian *Fagus sylvatica* forests (*Aremonio-Fagion*)	1.16
47	3160	Natural dystrophic lakes and ponds	1.12
48	3150	Natural eutrophic lakes with *Magnopotamion* or *Hydrocharition* - type vegetation	1.10
49	9170	*Galio-Carpinetum* oak-hornbeam forests	1.06
50	8310	Caves not open to the public	1.05
51	4030	European dry heaths	1.04
52	8130	Western Mediterranean and thermophilous scree	1.03
53	9560	*Endemic forests with *Juniperus* spp.	1.01
54	3130	Oligotrophic to mesotrophic standing waters with vegetation of the *Littorelletea uniflorae* and/or of the *Isoëto-Nanojuncetea*	1.00
55	9150	Medio-European limestone beech forests of the *Cephalanthero-Fagion*	0.98
56	3220	Alpine rivers and the herbaceous vegetation along their banks	0.97
57	8220	Siliceous rocky slopes with chasmophytic vegetation	0.94
58	3140	Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp.	0.91
59	3240	Alpine rivers and their ligneous vegetation with *Salix elaeagnos*	0.90
60	5110	Stable xerothermophilous formations with *Buxus sempervirens* on rock slopes (*Berberidion* p.p.)	0.86
61	9140	Medio-European subalpine beech woods with *Acer* and *Rumex arifolius*	0.85
62	9110	*Luzulo-Fagetum* beech forests	0.82
63	5130	*Juniperus communis* formations on heaths or calcareous grasslands	0.81
64	8240	*Limestone pavements	0.81
65	9340	*Quercus ilex* and *Quercus rotundifolia* forests	0.79
66	6170	Alpine and subalpine calcareous grasslands	0.79

Rank	Habitat code	Habitat name	Priority
67	9130	*Asperulo-Fagetum* beech forests	0.77
68	8210	Calcareous rocky slopes with chasmophytic vegetation	0.77
69	5210	Arborescent matorral with *Juniperus* spp.	0.75
70	8110	Siliceous scree of the montane to snow levels (*Androsacetalia alpinae* and *Galeopsietalia ladani*)	0.66
71	7120	Degraded raised bogs still capable of natural regeneration	0.66
72	6150	Siliceous alpine and boreal grasslands	0.64
73	8120	Calcareous and calcshist screes of the montane to alpine levels (*Thlaspietea rotundifolii*)	0.64
74	9120	Atlantic acidophilous beech forests with *Ilex* and sometimes also *Taxus* in the shrublayer (*Quercion robori-petraeae* or *Ilici-Fagenion*)	0.48
75	4090	Endemic oro-Mediterranean heaths with gorse	0.44
76	4060	Alpine and Boreal heaths	0.38

Table 3: Habitats of the Italian Continental biogeographical region ranked with the proposed method (* highlight priority habitats *sensu* Habitats Directive).

Rank	Habitat code	Habitat name	Priority
1	1320	*Spartina* swards (*Spartinion maritimae*)	3.08
2	1310	*Salicornia* and other annuals colonizing mud and sand	2.57
3	6420	Mediterranean tall humid grasslands of the *Molinio-Holoschoenion*	2.46
4	91B0	Thermophilous *Fraxinus angustifolia* woods	2.44
5	3170	*Mediterranean temporary ponds	2.34
6	1420	Mediterranean and thermo-Atlantic halophilous scrubs (*Sarcocornetea fruticosi*)	2.29
7	2230	*Malcolmietalia* dune grasslands	2.25
8	92A0	*Salix alba* and *Populus alba* galleries	2.23
9	7230	Alkaline fens	2.21
10	1410	Mediterranean salt meadows (*Juncetalia maritimi*)	2.21
11	7210	*Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae*	2.18
12	6210	Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (* important orchid sites)	2.13
13	2250	*Coastal dunes with *Juniperus* spp.	2.13
14	2260	*Cisto-Lavenduletalia* dune sclerophyllous scrubs	2.06
15	2110	Embryonic shifting dunes	1.96
16	9540	Mediterranean pine forests with endemic Mesogean pines	1.94
17	3240	Alpine rivers and their ligneous vegetation with *Salix elaeagnos*	1.92
18	2130	*Fixed coastal dunes with herbaceous vegetation ("grey dunes")	1.91
19	6520	Mountain hay meadows	1.84

Rank	Habitat code	Habitat name	Priority
20	1210	Annual vegetation of drift lines	1.83
21	91AA	*Eastern white oak woods	1.81
22	2120	Shifting dunes along the shoreline with *Ammophila arenaria* ("white dunes")	1.77
23	91F0	Riparian mixed forests of *Quercus robur, Ulmus laevis* and *Ulmus minor, Fraxinus excelsior* or *Fraxinus angustifolia*, along the great rivers (*Ulmenion minoris*)	1.73
24	9210	*Apennine beech forests with *Taxus* and *Ilex*	1.71
25	3250	Constantly flowing Mediterranean rivers with *Glaucium flavum*	1.67
26	3280	Constantly flowing Mediterranean rivers with *Paspalo-Agrostidion* species and hanging curtains of *Salix* and *Populus alba*	1.66
27	91E0	*Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion, Alnion incanae, Salicion albae*)	1.63
28	9260	*Castanea sativa* woods	1.63
29	2270	*Wooded dunes with *Pinus pinea* and/or *Pinus pinaster*	1.60
30	9220	*Apennine beech forests with *Abies alba* and beech forests with *Abies nebrodensis*	1.60
31	5310	*Laurus nobilis* thickets	1.57
32	3230	Alpine rivers and their ligneous vegetation with *Myricaria germanica*	1.53
33	5230	*Arborescent matorral with *Laurus nobilis*	1.50
34	6220	*Pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea*	1.48
35	9340	*Quercus ilex* and *Quercus rotundifolia* forests	1.46
36	62A0	Eastern sub-Mediterranean dry grasslands (*Scorzoneratalia villosae*)	1.46
37	7220	*Petrifying springs with tufa formation (*Cratoneurion*)	1.46
38	2160	Dunes with *Hippophaë rhamnoides*	1.42
39	6510	Lowland hay meadows (*Alopecurus pratensis, Sanguisorba officinalis*)	1.42
40	8240	*Limestone pavements	1.40
41	3150	Natural eutrophic lakes with *Magnopotamion* or *Hydrocharition* - type vegetation	1.40
42	3260	Water courses of plain to montane levels with the *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation	1.34
43	1340	*Inland salt meadows	1.28
44	7140	Transition mires and quaking bogs	1.27
45	9190	Old acidophilous oak woods with *Quercus robur* on sandy plains	1.27
46	6170	Alpine and subalpine calcareous grasslands	1.24
47	8220	Siliceous rocky slopes with chasmophytic vegetation	1.22
48	3130	Oligotrophic to mesotrophic standing waters with vegetation of the *Littorelletea uniflorae* and/or of the *Isoëto-Nanojuncetea*	1.21
49	9180	**Tilio-Acerion* forests of slopes, screes and ravines	1.17
50	6150	Siliceous alpine and boreal grasslands	1.14
51	5330	Thermo-Mediterranean and pre-desert scrub	1.11
52	4060	Alpine and Boreal heaths	1.07

Rank	Habitat code	Habitat name	Priority
53	4030	European dry heaths	1.06
54	3140	Hard oligo-mesotrophic waters with benthic vegetation of Chara spp.	1.05
55	5130	*Juniperus communis* formations on heaths or calcareous grasslands	1.04
56	1240	Vegetated sea cliffs of the Mediterranean coasts with endemic *Limonium* spp.	1.02
57	9410	Acidophilous *Picea* forests of the montane to alpine levels (*Vaccinio-Piceetea*)	0.98
58	6230	*Species-rich *Nardus* grasslands, on siliceous substrates in mountain areas (and submountain areas, in Continental Europe)	0.98
59	6110	*Rupicolous calcareous or basophilic grasslands of the *Alysso-Sedion albi*	0.97
60	2330	Inland dunes with open *Corynephorus* and *Agrostis grasslands*	0.94
61	8210	Calcareous rocky slopes with chasmophytic vegetation	0.93
62	91L0	Illyrian oak-hornbeam forests (*Erythronio-Carpinion*)	0.91
63	9160	Sub-Atlantic and medio-European oak or oak-hornbeam forests of the *Carpinion betuli*	0.90
64	6130	Calaminarian grasslands of the *Violetalia calaminariae*	0.90
65	6410	*Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (*Molinion caeruleae*)	0.90
66	9530	*(Sub-) Mediterranean pine forests with endemic black pines	0.78
67	5210	Arborescent matorral with *Juniperus* spp.	0.77
68	3110	Oligotrophic waters containing very few minerals of sandy plains (*Littorelletalia uniflorae*)	0.75
69	8310	Caves not open to the public	0.72
70	7150	Depressions on peat substrates of the *Rhynchosporion*	0.70
71	3220	Alpine rivers and the herbaceous vegetation along their banks	0.69
72	9120	Atlantic acidophilous beech forests with Ilex and sometimes also *Taxus* in the shrublayer (*Quercion robori-petraeae* or *Ilici-Fagenion*)	0.66
73	8130	Western Mediterranean and thermophilous scree	0.65
74	6430	Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels	0.61
75	9130	*Asperulo-Fagetum* beech forests	0.56
76	91M0	Pannonian-Balkanic turkey oak-sessile oak forests	0.54
77	9430	Subalpine and montane *Pinus uncinata* forests (* if on gypsum or limestone)	0.49
78	8230	Siliceous rock with pioneer vegetation of the *Sedo-Scleranthion* or of the *Sedo albi-Veronicion dillenii*	0.47
79	3270	Rivers with muddy banks with *Chenopodion rubri* p.p. and *Bidention* p.p. vegetation	0.46
80	8110	Siliceous scree of the montane to snow levels (*Androsacetalia alpinae* and *Galeopsietalia ladani*)	0.37
81	9110	*Luzulo-Fagetum* beech forests	0.35
82	5110	Stable xerothermophilous formations with *Buxus sempervirens* on rock slopes (*Berberidion* p.p.)	0.32

Rank	Habitat code	Habitat name	Priority
83	8120	Calcareous and calcshist screes of the montane to alpine levels (*Thlaspietea rotundifolii*)	0.32

Results are also summarized for groups of habitats that enabled a comparison of the priority values assigned to the habitats of the two biogeographical regions (Fig. 2). This showed that forests and grasslands had an overall high priority in both biogeographical regions. Bogs and fens are to be prioritized in the Alpine biogeograhical region while the coastal and dune habitats had high overall scores in the Continental biogeographical region. Rocks and screes habitats were of less concern for both regions.

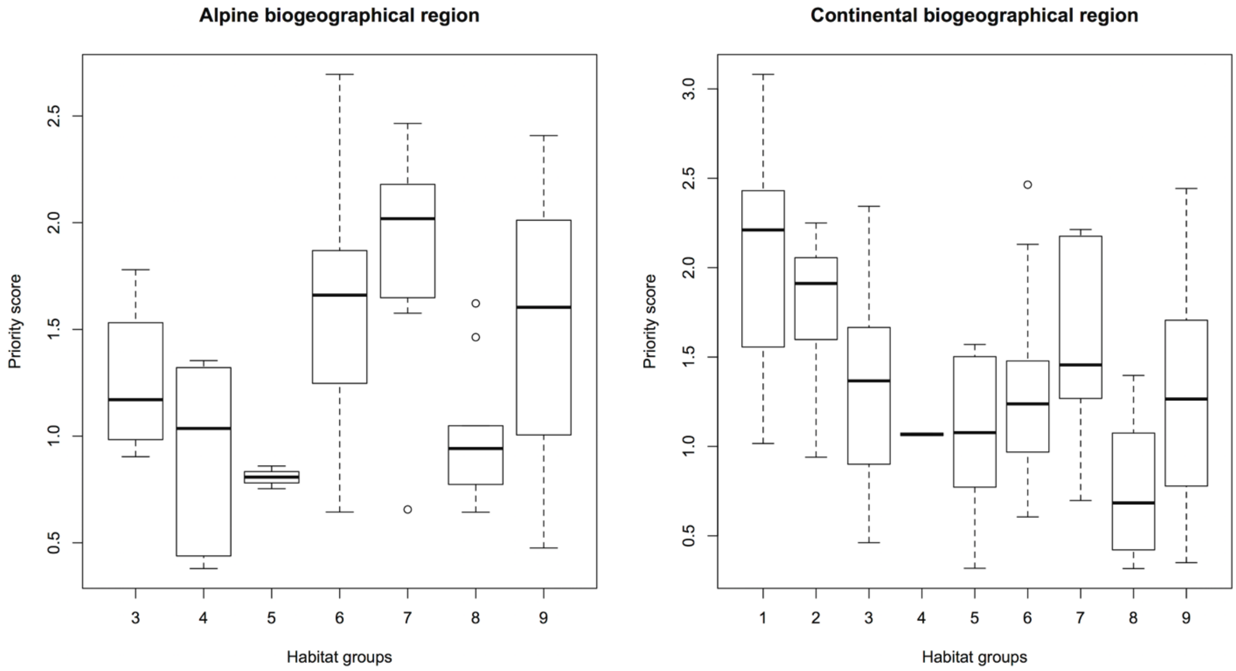

Figure 2: Priority scores of each habitat groups (1=coastal, 2= dune, 3= water, 4= heath, 5= scrubland, 6=grassland, 7=bogs, 8= rock and screes, 9= woodlands) for the two biogeographical regions

Four and six clusters were formed and identified for the Alpine and Continental biogeographical regions, respectively (Fig. 3). These groups do not represent single habitat categories (e.g., forest habitats) but habitats that may be completely different in ecological characteristics (e.g., 2260 and 91AA).

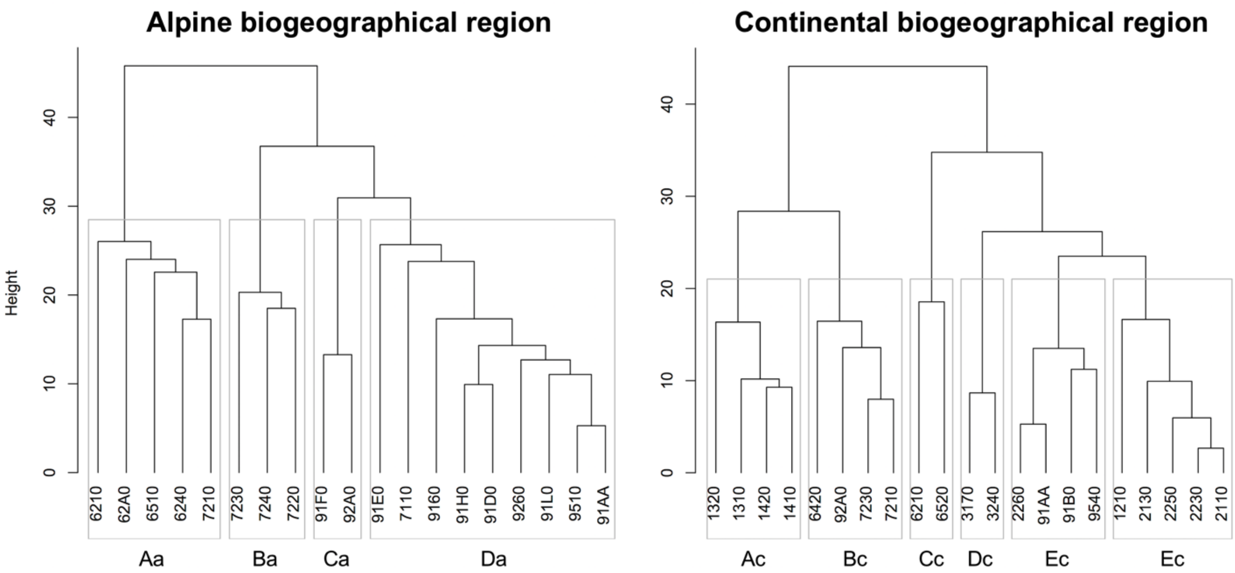

Figure 3: Dendrogram representing the four identified groups of habitats based on their affecting pressures for the Alpine, left, and Continental, right, biogeographical region

The analysis showed that several factors are shared and are important for the prioritized habitats of the Alpine and Continental biogeographical regions (Suppl. Material Fig. S1 and Table 4). Human induced changes in hydraulic conditions, agriculture and natural biotic and abiotic processes are those better representing the clusters (Table 4). However, common and most pressing affecting pressure to the prioritized habitats is “roads and motorways” (code D01.02) for both the biogeographical regions. “Other forestry activities” (code B07), (code D01.02), “improved access to site” (code D05), “outdoor sports, leisure and recreational activities” (code G01), and “vegetation succession/biocenotic evolution” (code K02) were most important for the Alpine region; whereas, “urbanisation and human habitation” (code E01) was common for the Continental region (Suppl. Material Fig. S1). Furthermore, among these, certain affecting pressures are shared by more habitat groupings; for example, “outdoor sports, leisure and recreational activities” and “urbanised areas, human habitation” for the Alpine and Continental regions, respectively (Table 4).

Table 4: Affecting pressures defining groups identified through cluster analysis and related indicator (stat) and p-values

Alpine group	Continental group	Affecting pressures	Definition	stat	p.value
	C	A02	Modification of cultivation practices	1	0.01
A		A02.01	Agricultural intensification	0.894	0.015
	C	A04.03	Abandonment of pastoral systems, lack of grazing	1	0.01
	F	A07	Use of biocides, hormones and chemicals	0.846	0.026

Alpine group	Continental group	Affecting pressures	Definition	stat	p.value
A		A08	Fertilisation	0.866	0.01
	B+F	A08	Fertilisation	0.816	0.044
C+D		B02	Forest and plantation management & use	0.853	0.018
C+D		B07	Other forestry activities	1	0.001
	C	C01	Mining and quarrying	1	0.01
	B+C+D+E	D01.02	Roads, motorways	0.894	0.013
B+C		D05	Improved access to site	0.852	0.035
	A+B+C+D	E01	Urbanisation and human habitation	0.931	0.004
A+C+D		G01	Outdoor sports, leisure and recreational activities	0.901	0.033
	C+E	G01.03	motorised vehicles	0.816	0.05
C		G05	Other human intrusions and disturbances	0.802	0.025
	D	G05.01	Trampling, overuse	0.91	0.002
	A+B+F	H01	Pollution to surface waters	0.866	0.036
C		I01	Invasive alien species	0.959	0.003
C		J02	Changes in water bodies conditions	0.934	0.003
	F	J.02.01.03	Infilling of ditches, dykes, ponds, pools, marshes or pits	0.943	0.021
	A	J.02.02.02	Estuarine and coastal dredging	1	0.002
	A+B	J02.03.02	Canalisation	0.935	0.002
	C+F	J02.05	Modification of hydrographic functioning, general	0.866	0.036
B		J02.06	Water abstractions from surface waters	0.816	0.024
	A+B	J02.07	Water abstractions from groundwater	1	0.001
C		J03	Other changes to ecosystems	0.926	0.003
A+C		K02	Vegetation succession/Biocenotic evolution	0.851	0.039

Alpine group	Continental group	Affecting pressures	Definition	stat	p.value
	C	K04.05	Damage by herbivores (including game species)	1	0.01
C		L08	Flooding (natural processes)	0.878	0.017

2.4 Discussion

2.4.1 A method to identifying habitat conservation priorities

Here we have presented a replicable approach that integrates data and information deriving from the regulatory requirements of the European Union to identify conservation priorities for natural and semi-natural habitats. This approach has enabled to consistently detect the main pressures requiring management for such habitats. Furthermore, our biogeographical perspective framed within national boundaries enables to take into account the importance of biogeographical peculiarities (Kukkala et al. 2016) and the intrinsic administrative character of management decisions.

This study considered conservation needs of natural and semi-natural habitats and impacting actions, features acknowledged as fundamental in setting conservation priorities (Stroud et al. 2014), and capitalizes from the existing methods proposed for setting conservation priorities (Bacchetta et al. 2012) with the availability of officially recognized data. Several properties of the proposed method have been described to be valuable such as limited number of criteria and no complex weighting system (Gauthier et al. 2010). We used and present score classes to evaluate the different criteria, a commonly applied procedure for defining conservation priorities (e.g., Gauthier et al. 2010; Bacchetta et al. 2012). Overall, the method ensures transparency, repeatability and consistency. Spatial information is used among other important information to achieve an appropriate ranking of habitats as highlighted with the “national responsibility” approach (Schmeller et al. 2008b). Indeed, one can integrate the “national responsibility” method with the conservation status (Schmeller et al. 2008a) and the applicability of this method in Europe has been widely discussed (Schmeller et al. 2014). In the proposed method, in addition to the cover relevance criteria, also the conservation condition considers spatial information as range and distribution data.

Management measures can be suggested based on the main and shared acting forces for the habitats of conservation priority. A focus on management measures is a priority aspect in nowadays conservation of biodiversity (Watson et al. 2014). Indeed, identifying and evaluating acting forces is important to derive and suggest conservation measures (e.g., Graziano et al. 2009). Grouping conservation management as in this approach enables understanding the urgency of measures (Velázquez et al. 2010) and retrieving the habitats that will benefit from their application and related area of application. Furthermore, while the Habitats Directive explicitly foresees management of Natura 2000 sites, the implementation of certain measures outside protected areas is also important for the overall conservation of habitats and their functional connectivity (Opermanis et al. 2013; Orlikowska et al. 2016).

The method is adaptable to future changes in data and in their availability. The information gathered within each criterion may be enlarged when other relevant information is obtainable. In the future it should be possible to consider other relevant parameters within the biodiversity value criteria. For example, among habitat typical species certain countries have presented list of animal species and these could be appropriately considered if applying our approach to these countries. For the same reason, scoring of this criterion may also take into account whether the habitat is within those entered in the "European Red List of Habitats Types" (Rodwell et al. 2013).

2.4.2 Habitat conservation priorities and management for the case study regions

Among the first top ranked habitats for conservation management in the Alpine biogeographical region there are bogs and fens, (dry) grasslands and forest habitats. These results are partially in line with previous research carried out for the Alpine areas. Bragazza (2009) highlighted high priority of conservation for rocky slopes and screes, peatlands, Alpine and arid grasslands, wet meadows and freshwater habitats. For the Continental biogegraphical region, a wider variety of habitats were ranked within the top ten to undergo conservation management: coastal and water habitats, dunes, fens, grasslands and forests. Indeed, human pressure is higher in the Continental compared to the Alpine region and this can be also evinced by the abovementioned variety.

Furthermore, to understand the goodness of the method, a simple comparison can be made with existing Prioritized Action Frameworks (PAFs) documents (see Article 8 (4) of the Habitats Directive) of Italian administrative regions falling within the Alpine and Continental biogeographical regions. For example, the PAF for the Veneto region (Veneto Region 2015; Causin et al. 2016), strongly based on expert opinion, showed congruence with our results highlighting several habitats (1210, 1410, 2110, 2130, 2230, 2250, 6210, 62A0, 7210, 9160, 91D0, 91E0, 92A0) among those to be prioritized for conservation management. Similarly, congruence on several habitats can be found from other regional PAFs. Covering the two biogeographical regions. Therefore, the proposed method could be useful as a tool for preparing PAFs.

Our study revealed management measures needed in the Alpine and Continental biogeographical regions. The importance of roads and motorways as affecting pressures in both regions indicate the need of specific management of traffic and transport systems. Interestingly, within the overall assessment at the EU level such pressure is not among the first ranking for habitats but it is for many species groups (EEA 2015). Several are the possible specific measures to be undertaken, among which: ecologically–sound planning and appropriate assessment of new roads (Sitzia et al. 2016a), identification of areas to keep as road-free and low-traffic (Selva et al. 2011) and of the spatial extent of roads impacts for habitats (see example for wildlife; Torres et al. 2016), and reducing effects through planning of restoration and mitigation measures (e.g. Ottburg and Blank 2015). Changes in agricultural activities with a polarization of these activities (i.e. intensification and abandonment) are an important factor in both biogeographical regions as also mentioned for many habitats in Europe (Ostermann 1998; Halada et al. 2011; Campagnaro et al. 2017) that must be tackled through appropriate policy (e.g. incentives) and management measures (e.g. mowing and grazing). Indeed, spontaneous succession phenomena and natural processes (e.g. flooding) are important in both areas and this should require active interventions to stabilize habitats not overlooking the natural dynamics of ecosystems. Water-related management measures, such as restoring and improving wetlands, water quality, hydrological regimes, and reducing the effects of abstractions, infilling and dredging will benefit a wide array of habitats in both regions (e.g., Verhoeven 2014; Sitzia et al. 2016c).

In addition, for the Alpine region there are several other conservation management measures that must be encouraged. Silviculture and forest exploitation must follow near-to-nature approaches considering habitats requirements, as it is widely occurring, and widespread abandonment of forestry activities is to be tackled, at least for certain habitats and areas, to enable preserve habitat heterogeneity. Similarly, outdoors sports, leisure and recreational activities must be carried out after suitable spatial planning avoiding direct impacts to habitats and species (e.g., Muñoz-Santos and Benayas 2012; Sitzia et al. 2014). Furthermore, the analysis revealed that also invasive alien species must be controlled through effective management measures in certain forest habitat types (Sitzia et al. 2016b). Other recommendations must be given for the Continental biogeographical region. The importance of urbanization and human habitation together with mining and quarrying give an indication that indiscriminate land use changes require appropriate spatial planning (Falcucci et al. 2007) with related mitigation and restoration action of the prioritized habitats.

2.4.3 Applicability and implications

This method can be useful even outside the European context where information on a combination of four criteria important for habitat conservation is available. Furthermore, while the approach is tested at the biogeographical level within national administrative boundaries, it can be possible to narrow or broaden its geographical application. Indeed, similar data is available at the Natura 2000 site level from the Standard Data Form but also in some cases by Natura 2000 Management Plans. For example, within the Standard Data Form information on the conservation degree of habitats and their cover is given, and main pressure and threats to the site are listed. In many cases also data from specific surveys are available that makes it possible to understand which species are locally linked to the habitats and whether there are species of conservation interest. Furthermore, an up scaling could be possible from data of each country that, through appropriate weights, could give indication at EU biogeographical level. Therefore, the application of this method could benefit decision makers at different levels and could be appropriate also for local managers constrained by limited funding availability. For example, it could be of great help in the design of Natura 2000 management plans, formulation of conservation measures, whether at a regional or site scale, definition of conservation objectives, and setting a strategy of action at the biogegraphical level. At the site level, our method integrated with other proposed methods (e.g., Caniani et al. 2016; Foresta et al. 2016) could be useful for appropriate assessment under Article 6(3) of the Habitats Directive of plans and projects. Furthermore, adequate adjustments could enable its application to prioritize conservation management of species (e.g. using population parameters).

This approach does not necessarily indicate habitats that should be tackled in view of the next monitoring period. However, the approach can be adapted to fulfill regulatory obligations and strategic objectives. In light of the EU 2020 Biodiversity Strategy, this approach will need modifications to rank as priorities those habitats requiring minimum action to achieve an increase in the conservation status. Therefore, the conservation condition and affecting pressures criteria will be used to understand which habitats should be considered in the first place and excluding those habitats already in a favorable conservation status.

Our method and, therefore, related results should be considered with relevant caveats. The most important is related to data availability and data quality. Indeed, the availability of biodiversity data is still recognized as a limit for conservation planning (Gaston et al. 2008; EEA 2015). While, the method to produce data under Article 17 of the Habitats Directive is reported, it must be emphasized that data can derive from expert knowledge, modeling applications, field surveys, and a mix of these. Therefore, there can be discrepancies and inconsistencies in the methodology between Member States (Schmeller et al. 2014). However, it can be assumed that a single Member State adopted a similar approach during the assessments. This information is increasingly been used in biodiversity conservation studies (e.g., Mazaris et al. 2013; Gigante et al. 2016), highlighting a degree of quality. Currently data from the Article 17 reporting is the most extensive database on the conservation status and distribution of habitats in the EU (Schmeller et al. 2014) and, as in the case of data on Natura 2000 sites, further adjustments to these databases will enable using more precise results (Duarte et al. 2016).

Nevertheless, we agree on the need of increasing scientific attention and on the inclusion of scientific data for achieving the requirements of Article 17 (Louette et al. 2015). Our approach focuses on conservation measures, but research efforts should focus on those habitats with a lack of data and information on the parameters forming our method. Furthermore, the method has a degree of subjectivity in the selection of the parameters and their scores that cannot be avoided. However, the legislative context limits this subjectivity by providing a set of specific features and this drawback could be further reduced with stakeholders inclusion in its application (Mikkonen and Moilanen 2013).

2.5 Conclusion

This method enabled to prioritize habitat management based on habitats conservation condition and value, the actions threating them, and their cover relevance at the relevant administrative area. The approach is specifically useful in the European Habitats Directive context and it can serve as a tool for tacking the necessary steps to avoid the deterioration of habitats under Article 6 of the Habitats Directive. The application of this method provides a list of habitats for which conservation measures are to be prioritized and enables to indicate conservation measures priorities with wider benefits for these habitats.

Improvements in data availability and coherence will benefit its application in Europe and in other contexts. The structure of this method enables its integration with new data deriving from scientific research, its application at different spatial scales important to conservation within Europe (i.e., EU-28 level, administrative regions, and site), and its testing outside Europe. This approach would enable transparent conservation decisions and has the potential to help policy makers and managers making conservation-sound decisions and achieving biodiversity targets.

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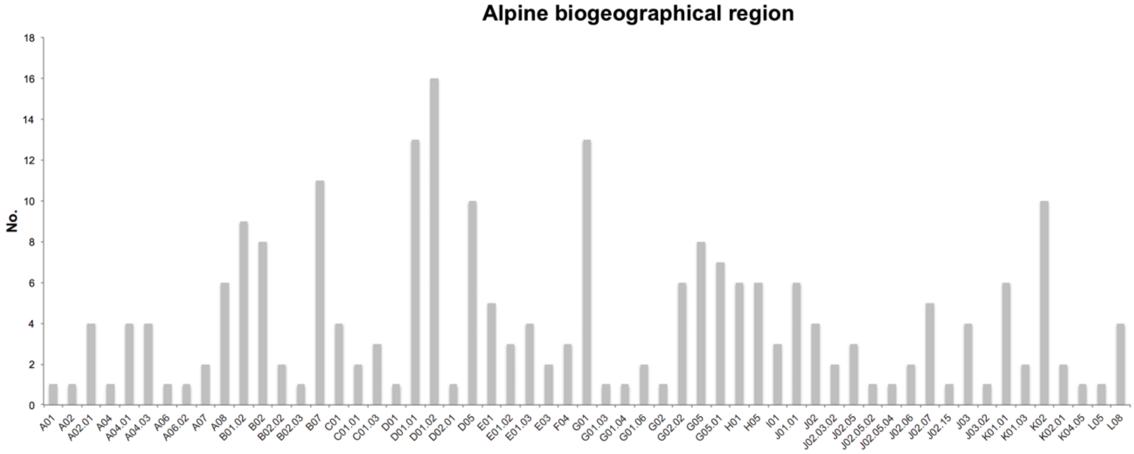
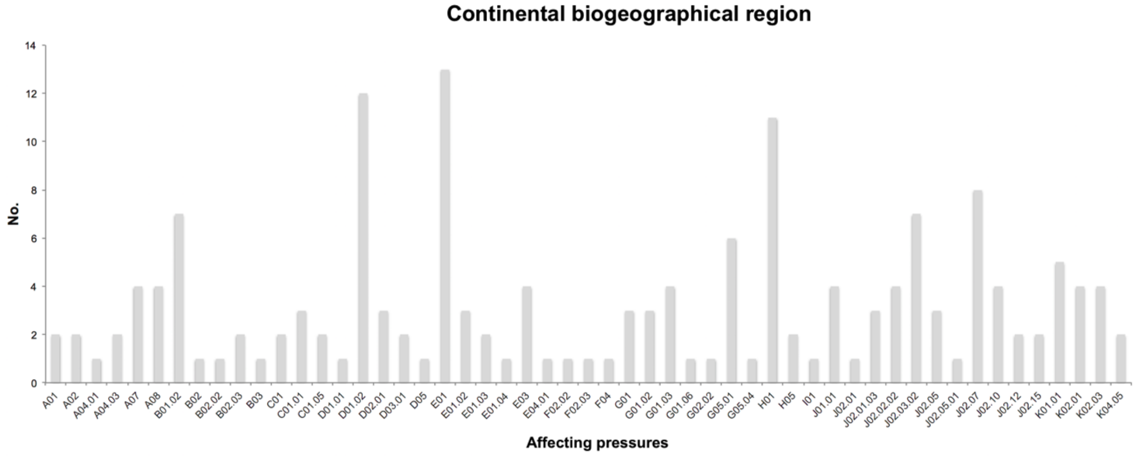
2.6 Supplementary Material

Figure 1S: Number of prioritized habitats (light grey) reported for each affecting pressure for the Alpine (top) and Continental (below) biogeographical regions. Codes refer to specific pressures as reported at http://bd.eionet.europa.eu/activities/Reporting/Article_17/reference_portal.





2.7 References

Angiolini C, Viciani D, Bonari G et al. (2016). Habitat conservation prioritization: A floristic approach applied to a Mediterranean wetland network. Plant Biosyst: 1-15. doi: 10.1080/11263504.2016.1187678

Bacchetta G, Farris E, Pontecorvo C (2012). A new method to set conservation priorities in biodiversity hotspots. Plant Biosyst 146: 37-41.

Bani L, Massimino D, Bottoni L et al. (2006). A multiscale method for selecting indicator species and priority conservation areas: a case study for broadleaved forests in Lombardy, Italy. Conserv Biol 20(2): 512-526. doi: 10.1111/j.1523-1739.2006.00331.x

Benavent-González A, Lumbreras A, Molina JA (2014). Plant communities as a tool for setting priorities in biodiversity conservation: a novel approach to Iberian aquatic vegetation. Biodivers Conserv 23(9): 2135-2154. doi: 10.1007/s10531-014-0709-3

Berg C, Abdank A, Isermann M et al. (2014). Red Lists and conservation prioritization of plant communities – a methodological framework. Appl Veg Sci 17: 504-515. doi: doi:10.1111/avsc.12093

Biondi E, Blasi C, Burrascano S et al. (2009). Manuale Italiano di interpretazione degli habitat della Direttiva 92/43/CEE. Società Botanica Italiana, Ministero dell'Ambiente e della Tutela del Territorio e del Mare, http://vnr.unipg.it/habitat.

Bragazza L (2009). Conservation priority of Italian Alpine habitats: a floristic approach based on potential distribution of vascular plant species. Biodivers Conserv 18: 2823-2835. doi: 10.1007/s10531-009-9609-3

Bunce RGH, Bogers MMB, Evans D et al. (2013). The significance of habitats as indicators of biodiversity and their links to species. Ecol Indic 33: 19-25. doi: 10.1016/j.ecolind.2012.07.014

Campagnaro T, Frate L, Carranza ML et al. (2017). Multi-scale analysis of alpine landscapes with different intensities of abandonment reveals similar spatial pattern changes: Implications for habitat conservation. Ecol Indic 74: 147-159. doi: 10.1016/j.ecolind.2016.11.017

Caniani D, Labella A, Lioi DS et al. (2016). Habitat ecological integrity and environmental impact assessment of anthropic activities: A GIS-based fuzzy logic model for sites of high biodiversity conservation interest. Ecol Indic 67: 238-249. doi: 10.1016/j.ecolind.2016.02.038

Causin L, Campagnaro T, Trentanovi G et al. (2016). Metodo e sintesi dei risultati ottenuti nella redazione del PAF "Prioritised Action Framework" per la conservazione della biodiversità nella rete Natura 2000 del Veneto. 7° Convegno Faunisti Veneti. Bonato L, Trabucco R, Bon M. Bollettino del Museo di Storia Naturale di Venezia, Verona, Italy. 66**:** 9-19.

D'Amen M, Bombi P, Campanaro A et al. (2013). Protected areas and insect conservation: questioning the effectiveness of Natura 2000 network for saproxylic beetles in Italy. Anim Conserv 16: 370-378. doi: 10.1111/acv.12016

De Cáceres M, Legendre P, Moretti M (2010). Improving indicator species analysis by combining groups of sites. Oikos 119: 1674-1684. doi: 10.1111/j.1600-0706.2010.18334.x

Duarte I, Rego FC, Casquilho JP et al. (2016). A Relevance Index for the habitat areas of Natura 2000 Network based on their Rarity and Representativeness. Ecol Indic 61: 202-213. doi: 10.1016/j.ecolind.2015.09.015

Dufrêne M, Legendre P (1997). Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol Monogr 67: 345-366. doi: 10.2307/2963459

EC (2011). Our life insurance, our natural capital: an EU biodiversity strategy to 2020. European Commission, Brussels.

EC (2013). Interpretation manual of European union habitats. European Commissione DG Environment, Luxembourg.

EEA (2012). Protected areas in Europe: an overwiev. EEA Report Nº 5/2012. European Environment Agency, Copenhagen.

EEA (2015). State of nature in the EU. Results from reporting under the nature directives 2007-2012. EEA Technical report No 2/2015. European Environment Agency, Copenhagen.

Evans D (2006). The habitats of the European Union Habitats Directive. Biol Environ-Proc R Irish Acad 106B: 167-173.

Evans D, Arvela M (2011). Assessment and Reporting under Article 17 of the Habitats Directive–Explanatory Notes & Guidelines for the Period 2007-2012. Final Draft, July 2011. European Topic Centre on Biological Diversity, Paris.

Fabbris L (1997). Statistica multivariata. Analisi esplorativa dei dati. McGraw-Hill Libri Italia, Milano.

Falcucci A, Maiorano L, Boitani L (2007). Changes in land-use/land-cover patterns in Italy and their implications for biodiversity conservation. Landsc Ecol 22: 617-631. doi: 10.1007/s10980-006-9056-4

Fenu G, Fois M, Cogoni D et al. (2015). The Aichi Biodiversity Target 12 at regional level: an achievable goal? Biodivers 16: 120-135.

Foresta M, Carranza ML, Garfì V et al. (2016). A systematic conservation planning approach to fire risk management in Natura 2000 sites. J Environ Manage 181: 574-581. doi: 10.1016/j.jenvman.2016.07.006

Gaston KJ, Jackson SF, Nagy A et al. (2008). Protected Areas in Europe. Ann N Y Acad Sci 1134: 97-119. doi: 10.1196/annals.1439.006

Gauthier P, Debussche M, Thompson JD (2010). Regional priority setting for rare species based on a method combining three criteria. Biol Conserv 143: 1501-1509.

Gauthier P, Foulon Y, Jupille O et al. (2013). Quatifying vulnerability to assess priorities for conservation management. Biol Conserv 158: 321-325.

Geijzendorffer IR, Regan EC, Pereira HM et al. (2015). Bridging the gap between biodiversity data and policy reporting needs: an essential biodiversity variables perspective. J Appl Ecol 53: 1341-1350. doi: 10.1111/1365-2664.12417

Genovesi P, Angelini P, Bianchi E et al. (2014). Specie e habitat di interesse comunitario in Italia: distribuzione, stato di conservazione e trend. ISPRA, Rome.

Gigante D, Foggi B, Venanzoni R et al. (2016). Habitats on the grid: The spatial dimension does matter for red-listing. J Nat Conserv 32: 1-9.

Graziano R, Parolo G, Ferrarini A (2009). A rapid and cost-effective tool for managing habitats of the Europena Natura 2000 network: a case study in the Italian Alps. Biodivers Conserv 18: 1375-1388.

Halada L, Evans D, Romão C et al. (2011). Which habitats of European importance depend on agricultural practices? Biodivers Conserv 20: 2365-2378. doi: 10.1007/s10531-011-9989-z

Hochkirch A, Schmitt T, Beninde J et al. (2013). Europe needs a new vision for a Natura 2000 Network. Conserv Lett 6: 462-467. doi: 10.1111/conl.12006

Jantke K, Schleupner C, Schneider UA (2011). Gap analysis of European wetland species: priority regions for expanding the Natura 2000 network. Biodivers Conserv 20: 581-605. doi: 10.1007/s10531-010-9968-9

Kukkala AS, Arponen A, Maiorano L et al. (2016). Matches and mismatches between national and EU-wide priorities: examining the Natura 2000 network in vertebrate species conservation. Biol Conserv 198: 193-201. doi: 10.1016/j.biocon.2016.04.016

Legendre P, Legendre L (1998). Numerical Ecology. Elsevier Science, Amsterdam.

Louette G, Adriaens D, Adriaens P et al. (2011). Bridging the gap between the Natura 2000 regional conservation status and local conservation objectives. J Nat Conserv 19: 224-235.

Louette G, Adriaens D, Paelinckx D et al. (2015). Implementing the Habitats Directive: How science can support decision making. J Nat Conserv 23: 27-34. doi: 10.1016/j.jnc.2014.12.002

Maiorano L, Amori G, Montemaggiori A et al. (2015). On how much biodiversity is covered in Europe by national protected areas and by the Natura 2000 network: insights from terrestrial vertebrates. Conserv Biol 29: 986-995.

Maiorano L, Falcucci A, Boitani L (2006). Gap analysis of terrestrial vertebrates in Italy: Priorities for conservation planning in a human dominated landscape. Biol Conserv 133: 455-473.

Maiorano L, Falcucci A, Garton EO et al. (2007). Contribution of the Natura 2000 network to biodiversity conservation in Italy. Conserv Biol 21: 1433-1444.

Martín JL, Cardoso P, Archavaleta M et al. (2010). Using taxonomically unbiased criteria to prioritize resource allocation for oceanic island species conservation. Biodivers Conserv 19: 1659-1682.

Mazaris AD, Papanikolaou AD, Barbet-Massin M et al. (2013). Evaluating the connectivity of a protected areas' network under the prism of global change: the efficiency of the European Natura 2000 network for four birds of prey. PLoS ONE 8: e59640. doi: 10.1371/journal.pone.0059640

Mikkonen N, Moilanen A (2013). Identification of top priority areas and management landscapes from a national Natura 2000 network. Environ Sci Policy 27: 11-20. doi: 10.1016/j.envsci.2012.10.022

Muñoz-Santos M, Benayas J (2012). A proposed methodology to assess the quality of public use management in protected areas. Environ Manage 50: 106-122. doi: 10.1007/s00267-012-9863-0

Opermanis O, MacSharry B, Evans D et al. (2013). Is the connectivity of the Natura 2000 network better across internal or external administrative borders? Biol Conserv 166: 170-174. doi: 10.1016/j.biocon.2013.06.019

Orlikowska EH, Roberge J-M, Blicharska M et al. (2016). Gaps in ecological research on the world's largest internationally coordinated network of protected areas: a review of Natura 2000. Biol Conserv 200: 216-227. doi: 10.1016/j.biocon.2016.06.015

Ostermann OP (1998). The need for management of nature conservation sites designated under Natura 2000. J Appl Ecol 35: 968-973. doi: 10.1111/j.1365-2664.1998.tb00016.x

Ottburg F, Blank M (2015). Solutions to the impacts of roads and other barriers on fish and fish habitat. In: van der Ree R, Smith DJ, Grilo C (Eds) Handbook of road ecology. John Wiley & Sons Ltd, West Sussex, 364-373.

Popescu VD, Rozylowicz L, Cogălniceanu D et al. (2013). Moving into Protected Areas? Setting conservation priorities for Romanian reptiles and amphibians at risk from climate change. PLoS ONE 8: e79330. doi: 10.1371/journal.pone.0079330

Popescu VD, Rozylowicz L, Niculae IM et al. (2014). Species, habitats, society: an evaluation of research supporting EU’s Natura 2000 network. PLoS ONE 9: e113648. doi: 10.1371/journal.pone.0113648

Pullin AS, Báldi A, Can OE et al. (2009). Conservation focus on Europe: major conservation policy issues that need to be informed by conservation science. Conserv Biol 23: 818-824.

R Development Core Team (2015). R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Richard D, Bailly Maitre J, Aronsson M et al. (2016). Supporting elements for the Atlantic Natura 2000 review seminar (1st part: Core document). ETC/BD report to the EEA. European Topic Centre on Biological Diversity, Paris, France.

Rodwell J, Janssen J, Gubbay S et al. (2013). Red List assessment of European habitat types—A feasibility study. Service Contract No. 070307/2012/624047/SER/B3, Report for the European Commission, DG Environment.

Rossi G, Montagnani C, Gargano D et al. (2013). Lista Rossa della flora italiana. Policy species e altre specie minacciate. Comitato Italiano IUCN e Ministero dell'Ambiente e della Tutela del Territorio e del Mare, Rome.

Salafsky N, Salzer D, Stattersfield AJ et al. (2008). A standard lexicon for biodiversity conservation: unified classification of threats and actions. Conserv Biol 22: 897-911.

Schatz B, Gauthier P, Debussche M et al. (2014). A decision tool for listing species for protection on different geographic scales and administrative levels. J Nat Conserv 22: 75-83. doi: 10.1016/j.jnc.2013.09.003

Schmeller DS, Bauch B, Gruber B et al. (2008a). Determination of conservation priorities in regions with multiple political juristictions. Biodivers Conserv 17: 3623-3630.

Schmeller DS, Evans D, Lin Y-P et al. (2014). The national responsibility approach to setting conservation priorities – recommendations for its use. J Nat Conserv 22: 349-357.

Schmeller DS, Gruber B, Bauch B et al. (2008b). Determination of national conservation responsabilities for species conservation in regions with multiple political juristictions. Biodivers Conserv 17: 3607-3622.

Schmeller DS, Maier A, Evans D et al. (2012). National responsibilities for conserving habitats – a freely scalable method. Nature Conserv 3: 21-44. doi: 10.3897/natureconservation.3.3710

Schnittler M, Günther K-F (1999). Central European vascular plants requiring priority conservation measures – an analysis from national Red Lists and distribution maps. Biodivers Conserv 8: 891-925. doi: 10.1023/a:1008828704456

Selva N, Kreft S, Kati V et al. (2011). Roadless and low-traffic areas as conservation targets in Europe. Environ Manage 48(5): 865-877. doi: 10.1007/s00267-011-9751-z

Sitzia T, Campagnaro T, Grigolato S (2016a). Ecological risk and accessibility analysis to assess th eimpact of roads under Habitats Directive. J Environ Plan Manage 59: 2251-2271. doi: 10.1080/09640568.2016.1140023

Sitzia T, Campagnaro T, Kowarik I et al. (2016b). Using forest management to control invasive alien species: helping implement the new European regulation on invasive alien species. Biol Invasions 18: 1-7. doi: 10.1007/s10530-015-0999-8

Sitzia T, Michielon B, Iacopino S et al. (2016c). Population dynamics of the endangered shrub *Myricaria germanica* in a regulated Alpine river is influenced by active channel width and distance to check dams. Ecol Eng 95: 828-838. doi: 10.1016/j.ecoleng.2016.06.066

Sitzia T, Rizzi A, Cattaneo D et al. (2014). Designing recreational trails in a forest dune habitat using least-cost path analysis at the resolution of visitor sight distance. Urban For Urban Greening 13: 861-868. doi: 10.1016/j.ufug.2014.09.011

Stroud JT, Rehm E, Ladd M et al. (2014). Is conservation research money being spent wisely? Changing trends in conservation research priorities. J Nat Conserv 22: 471-473. doi: 10.1016/j.jnc.2014.05.003

Sundseth K, Creed P (2008). Natura 2000: Protecting Europe's Biodiversity. Office for Official Publications of the European Communities, Luxembourg.

Tomaselli V, Dimopoulos P, Marangi C et al. (2013). Translating land cover/land use classifications to habitat taxonomies for landscape monitoring: a Mediterranean assessment. Landsc Ecol 28: 905-930. doi: 10.1007/s10980-013-9863-3

Torres A, Jaeger JAG, Alonso JC (2016). Assessing large-scale wildlife responses to human infrastructure development. Proc Natl Acad Sci USA 113: 8472-8477. doi: 10.1073/pnas.1522488113

Tsiafouli MA, Apostolopoulou E, Mazaris AD et al. (2013). Human activities in Natura 2000 sites: a highly diversified conservation network. Environ Manage 51: 1025-1033. doi: 10.1007/s00267-013-0036-6

Velázquez J, Tejera R, Hernando A et al. (2010). Environmental diagnosis: integrating biodiversity conservation in management of Natura 2000 forest spaces. J Nat Conserv 18: 309-317. doi: 10.1016/j.jnc.2010.01.004

Veneto Region (2015). Programma di Sviluppo Rurale per il Veneto 2007-2013. DGR n. 746 del 15 marzo 2010 e s.m.i., Misura 511- Assistenza tecnica. Approvazione del documento "Prioritised Action Framework - PAF" per le aree nella Rete Natura 2000 relativamente al periodo di programmazione comunitaria 2014-2020. Deliberazione della Giunta Regionale n. 683.

Verhoeven JTA (2014). Wetlands in Europe: perspectives for restoration of a lost paradise. Ecol Eng 66: 6-9. doi: 10.1016/j.ecoleng.2013.03.006

Votsi N-EP, Zomeni MS, Pantis JD (2016). Evaluating the effectiveness of Natura 2000 network for wolf conservation: a case-study in Greece. Environ Manage 57: 257-270. doi: 10.1007/s00267-015-0621-y

Watson JEM, Dudley N, Segan DB et al. (2014). The performance and potential of protected areas. Nature 515: 67-73. doi: 10.1038/nature13947

Zhang L, Xu W-h, Ouyang Z-y et al. (2014). Determination of priority nature conservation areas and human disturbances in the Yangtze River Basin, China. J Nat Conserv 22: 326-336. doi: 10.1016/j.jnc.2014.02.007

3. Paper II: Ecological risk and accessibility analysis to assess the impact of roads under Habitats Directive[2]

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Abstract

We propose a method for the appropriate assessment of adverse effects of roads in compliance with the European Union Habitats Directive. The method incorporates an analysis of ecological risk of edge effects by the proposed roads with the related increase in accessibility. The method was tested on 30 km of planned forest roads inside an 8,000-ha reserve included in two Natura 2000 sites. As a result, the cumulative effect of 19 road segments was judged as not significantly affecting the integrity of the sites, although they made accessible an extra 314 ha. On the basis of the accessibility calculation, 20 ha of land were set aside from forest exploitation as a mandatory mitigation measure. The method objectively determined the cumulative adverse effects, enabled comparison of plan revisions and alternatives and proved to measure direct and indirect significant effects with a realistic effort in terms of field survey and geographic information system processing.

Keywords: Environmental Impact Assessment; Natura 2000; biodiversity conservation; forest management; appropriate impact assessment

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3.1 Introduction

Sites of Community Importance (SCIs) or Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) form the network of Natura 2000 sites in Europe. These sites are selected based on Annexes I and II (list of habitats and wild flora and fauna, respectively) of the Habitats Directive (92/43/EEC) and Annex I (list of birds) of the Birds Directive (2009/147/EC). Appropriate assessment (AA) is a process required by Article 6(3) of the Habitats Directive for plans and projects that are likely to have a significant adverse effect in relation to the conservation objectives of Natura 2000 sites. Member states and regional bodies have incorporated this requirement into national and regional legislation. Awareness of the need for AA has only recently been raised among actors (Beunen 2006) and the process through which assessments are made is still an important aspect for the maintenance of Natura 2000 sites in different member states (Ferranti, Beunen, and Speranza 2010). The need for AA has caused the delay of planning processes that in several cases ended in judicial intervention (Beunen 2006). This process, particularly the assessment of plans, presents some reviewed experiences of its application (Söderman 2009; Therivel 2009). General (European Commission 2000a, 2002, 2012a) and specific (European Commission 2010, 2011a, 2011b, 2012b, 2012c) documents with methodological indications on AA have been published by the European Union, together with other documents published by national bodies, consulting agencies and NGOs (e.g., BMVI 2004; Scott Wilson, Levett-Therivel Sustainability Consultants, Treweek Environmental Consultants, and Land Use Consultants 2006). Much attention has been given to legal and procedural aspects, overlooking the environmental outcomes and the applied methods (Beunen, van der Knaap, and Biesbroek 2009). Conservation scientists involved in the implementation of Natura 2000 have underlined the need to improve the quality of these assessments (Kati et al. 2015).

One fundamental step of AA is the risk estimation of an impact brought by projects and plans on the integrity of a protected site (Opdam, Broekmeyer, and Kistenkas 2009). The European Commission suggests assessing the significance of impacts “upon factors such as the perceived value of the affected environment, the magnitude, spatial extent and duration of anticipated change; the resilience of the environment to cope with change” (European Commission 2002, 62). Indeed, the identification of the ecological risk (Landis and Wiegers 1997) caused by a plan or a project is in accordance with the above-mentioned statement. The approach presented in this paper analyses hazards, vulnerability and specific ecological values. It, in fact, evaluates the probability of potential negative impacts derived from the exposure to one or more stress factors on different organisational scales (Suter 1993). The analysis of risk enables the improvement of environmental assessment and protection, to make more rational decisions shifting away from subjectivity (Suter, Barnthouse, and O’Neill 1987) and to solve complex ecological problems (Lackey 1998). Furthermore, ecological risk assessment has a wide applicability, as it is able to take into account a variety of impacts and ecosystem components (Fock, Kloppman, and Stelzenmüller 2011). These distinctive features of ecological risk (e.g., the consideration of ecological values, the magnitude of stress factors, the wide applicability) allow including the factors mentioned by the European Commission for the assessment of significance. However, up to now, ecological risk assessment has been applied by a narrow group of researchers (Dana, Kapuscinski, and Donaldson 2012) and rarely by practitioners.

Guidelines on AA (e.g., European Commission 2002) stress the importance of spatially defining the assessment. However, spatially explicit information is not always presented and analysed. Söderman (2009) studied AA in Finland and observed that the study area was identified mainly with a non-ecological approach, around 70% of the studies reported, in a map, the plan or project site, and only 30%40% of the studies graphically localised habitats and habitats of species. Furthermore, the author highlights that maps are fundamental tools to convey assessment results to decision-makers. Indeed, spatial identification and analysis through GIS tools has an important informative potential in the assessment of impacts by human activities on biodiversity (e.g., Scolozzi and Geneletti 2011). However, these tools are poorly used in ecological impact assessment (Gontier, Balfors, and Mörtberg 2006).

A recent study on Natura 2000 (European Environment Agency 2015) underlined that more than 60% of the assessments of forest and woodland habitats and species listed in the Habitats Directive revealed a bad or inadequate conservation status and that roads are among the most important pressures and threats to these habitats and species. However, forest roads are important for an economically viable forest management (European Commission 2015a) and to apply frequent and careful conservation management activities. Forest roads may have many complex effects on biotic and abiotic factors in woodland ecosystems, over time and space (Coffin 2007; Forman et al. 2003; Gucinski et al. 2001; Lugo and Gucinski 2000; Robinson, Duinker, and Beazley 2010; Spellerberg 1998; Trombulak and Frissel 2000).

Ecosystem components are directly and indirectly affected by roads over variable spatial extents (Forman 2000). Several authors have highlighted different extents of direct edge effects on understory plant (e.g., Avon et al. 2010; Haskell 2000; Watkins et al. 2003) and animal communities (Benítez-López, Alkemade, and Verweij 2010). By reducing the time that operators need to reach an extraction area (Hippoliti 1976), forest roads enlarge the area accessible for timber extraction and subject to indirect human disturbance. The area made accessible by the forest road network can be investigated based on the terrain morphology and on the extraction techniques and costs (Cavalli and Grigolato 2009; Zambelli et al. 2012). In this context, timber extraction efficiency can be evaluated by considering the overlapping effect of forest roads that, as a result, minimise the marginal effect derived from constructing new roads (Pentek et al. 2005; Ghaffariyan, Stampfer, and Sessions 2010) and increase the cumulative environmental impacts of new roads in respect to the existing ones (Gumus, Acar, and Toksoy 2008; Hayati, Majnounian, and Abdi 2012). To analyse the spatial extent of road effects on biodiversity, geographic information systems (GISs) are currently necessary and powerful tools (Karlson and Mörtberg 2015).

Here we propose and test a method that evaluates and controls the effects of forest road plans, their revisions and alternative solutions, on the integrity of Natura 2000 sites. The aim is to explain the contents of this method and to assess its effectiveness. Furthermore, we aim at identifying possible negative effects of the plan on species and habitats and at identifying the area made accessible by the forest roads. The method is replicable, not excessively complex while still being objective and it relates the ecological risk to the direct edge road effects and the forest accessibility to the indirect road effects. We test the proposed method through the use of a 30-km forest road plan case study from an 8,000-ha reserve in a dolomitic nature park that is included in two Natura 2000 sites. We then discuss the results in view of the current practice, guidelines by the European Commission and ruling by the European Court of Justice (ECJ). Our study contributes to the need for more clear approaches to improve the quality of AA (Söderman 2009; Therivel 2009), in particular for those assessments linked to small-scale activities (Kati et al. 2015).

3.2 Case Study

3.2.1 Study area and geographical context

To test the approach we selected a forest road plan within the Adamello Brenta Nature Park boundaries in the province of Trento (North-East Italy) (Figure 1 (a) and 1(b)). This plan underwent AA of impacts and covered an area that is considered large enough to be representative of a variety of site conditions commonly encountered in other similar cases. The area is included in two Natura 2000 sites, the SCI “IT3120177 – Dolomiti di Brenta” and the SPA “IT3120159 – Brenta.” The sites’ conservation objectives include the maintenance of habitats of high conservation interest, such as mixed silver fir forests, and of many alpine animal species, which include raptors and forest grouse. In general, the forests at these sites display a very high degree of wilderness (see European Environment Agency 2010a, 2010b). The surface included in the forest road plan is 8,155 ha and represents a reserve zone of the park. The area was a special reserve for the conservation of brown bears (*Ursus arctos* L.) and, to prevent the local extinction of this species, a LIFE project led to the reintroduction of several individuals in 1999. Now, the bear population is expanding and the original objective of the special reserve is no longer operative. For this reason, the Park authorities decided, in 2003, to allow municipalities to resume the controlled construction of forest roads, which are needed to apply extensive silvicultural and farming management techniques, i.e., frequent, moderate and careful treatments. However, the Park authorities decided that the planning of new forest roads should be done on a multi-municipality level to ensure the coherent and sustainable management of forests and pastures, thus providing the opportunity to test our method on one of the first examples of AA of an extended forest road network in Europe.

3.2.2 The forest road plan

The first plan was developed by the Provincial Forest Service in 2006 and included 26 proposed roads for a total of ca. 30 km (Figure 1). The main function of these roads was timber extraction. The related AA rejected 8 of the 26 proposed roads because of their likely negative impact on the sites’ integrity. According to the results of this AA, the Park approved the first forest road plan in 2008. In 2012, a revision of this plan was then forwarded by the Park and submitted for AA, with the consensus of the municipalities. The

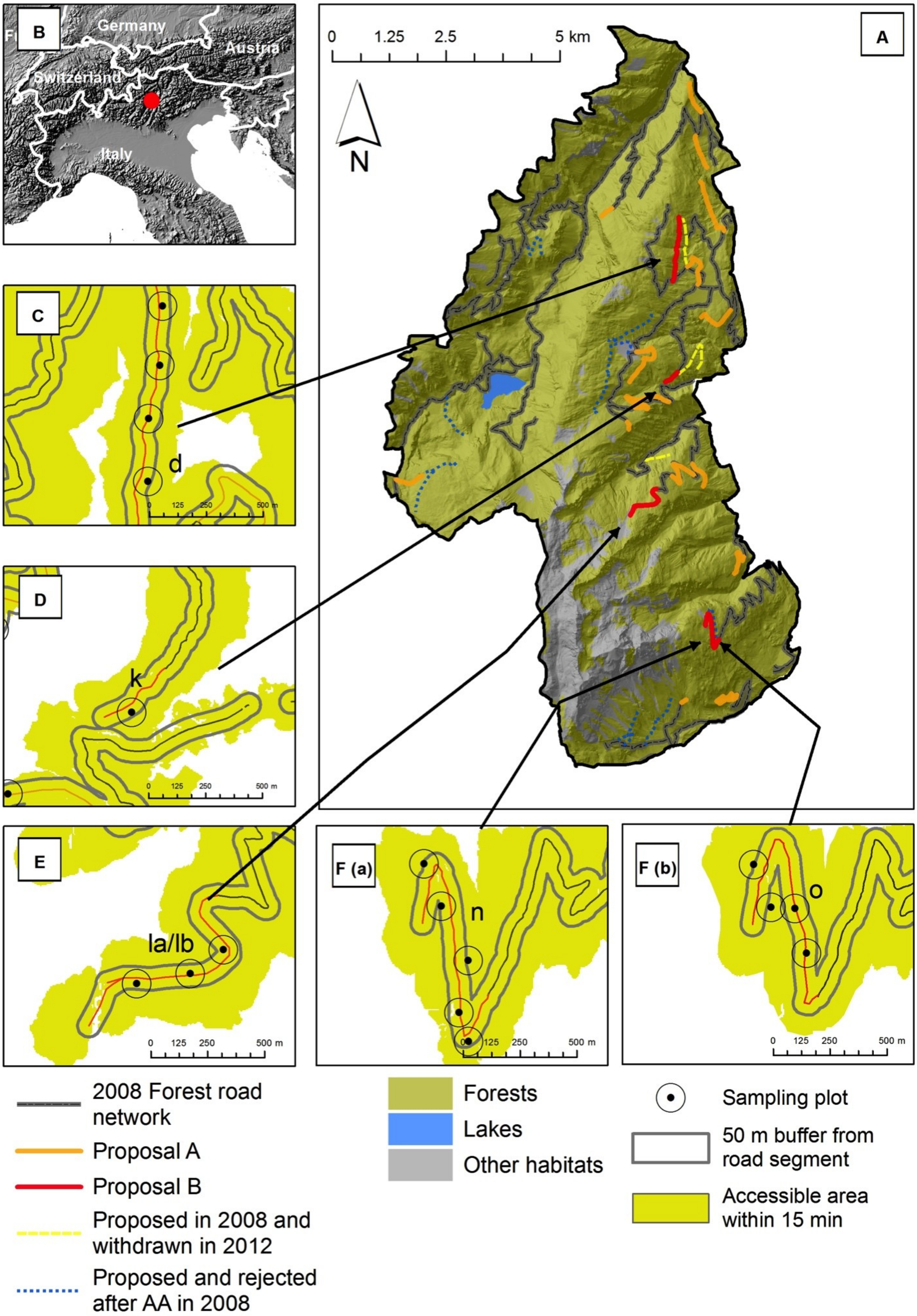

Figure 1: The forest road plan where the appropriate assessment was conducted. The study area with the forest roads (a), location of the study area (b), maps of the new roads within the proposal B (c–f), the initially proposed road n (*F*(*a*)) and its alternative o (*F*(*b*)).

revision contained 10 priority roads derived from the first version of the plan, i.e., those roads neither rejected after the AA in 2008 nor withdrawn. In addition, it included the proposal for four new priority roads, one of which with two alternatives (coded *Ia/Ib*) (Figure 1(c)–(f)). This revision was a consequence of the updating of several forest management plans and of the current necessity of tackling the abandonment of pasture lands in this region (Sitzia and Trentanovi 2011). This necessity mirrored the aim of the Nature Park in maintaining and promoting traditional culture, practices, landscapes and biodiversity. To test our method we assessed the latest version of the plan.

3.3 Methodology

3.3.1 Conceptual framework

According to the Habitats Directive, the AA should ascertain that no significant adverse effects impact the integrity of a protected site. This is based on the precautionary principle (European Commission 2002), which implies that scientific evaluation should aim at determining, with sufficient certainty, the risk encompassed in a plan or project (European Commission 2000b).

The identification of the area potentially affected is an important aspect for all projects and plans undergoing AA. Any AA of a forest road plan should always identify a spatially explicit area being probably affected by the related disturbances. Inside this area, the probability of significant adverse effects should decrease with increasing distance from the roads, following a probability density function. Beyond a certain distance, the probability of an effect should become very low. This distance identifies where the road edge effects take place, the road effect zone (Forman et al. 1997). In the absence of empirical site-specific studies on the size of this distance, the only solution is to use data from the existing literature, even if the identification of this area is considered controversial because of the difficulty of precisely defining the edge itself (Ries et al. 2004). However, the significance of direct effects could be considered negligible beyond a maximum distance from the road; therefore, allowing the establishment of critical thresholds within which the risk of significant effects due to road construction and use is probable (Opdam, Broekmeyer, and Kistenkas 2009).

The methodology that is applied and proposed is as follows:

(1) Identification of the ecological risk.

  (a) Identification of the pressures and threats and, therefore, of the hazard linked to the plan.

  (b) Identification of species and habitats occurring in the area of influence of the plan and their related value.

  (c) Identification of the vulnerability of identified habitats and species to the pressure and threats determined at point 1.1.

(2) Identification of areas made accessible by new road construction.

(3) Another two steps can be applied when needed, as was the case for this study, i.e., when alternative solutions and mitigations are provided by the plan to make it accordant with the site(s).

(4) Assessment of alternative solutions.

(5) Definition of mitigation actions.

3.3.2 Direct effects related to ecological risk

Assessing the possible risks of direct adverse effects caused by a forest road plan on Natura 2000 sites requires a tool that is able to appraise the potential disturbances on the nature values of a complex and dynamic ecosystem. Any risk (*R*) is a composite of the probability and magnitude of these undesirable outcomes. It is connected to the loss of value that a system and its components can suffer depending on the occurrence of possible negative phenomena within a defined physical, ecological, social, economic, cultural, political and legal administrative context (Lawrence 2007a).

We expressed risk as follows:

$$R = f(\text{hazard, value, vulnerability}) \tag{1}$$

This function relates risk to (1) the hazard expressing the probability of damage caused by the plan under investigation, (2) the value indicating the potential damage to the ecosystem, based on units that indicate its quality, and (3) the vulnerability (or sensitivity) indicating the predisposition of the ecosystem to experience damage. Equation (1) is identical to the equation of total risk in landslide hazard zonation, which is cited in a large number of publications, applied in many national regulations and was reviewed at a global level by Varnes (1984).

Dividing each value x_i by the largest observed value x_{max} is a way to have all values in the range [0,1] (Cain and Harrison [1958] as cited in Legendre and Legendre [1998]). Therefore, the scaled value $x_i^{'} \in [0,1] = x_i \cdot \rho_{x_i}$ where:

$$\rho_{x_i} = \frac{1}{x_{\max}} \tag{2}$$

Hence, the scaled value will be the proportion of x_i compared to x_{max}. This latter value (x_{max}) can be derived from all the values observed in a representative and vast enough area, as well as from regulatory thresholds or project-specific criteria (Lawrence 2007b). Before entering the formulas, original values have to be scaled with Equation (2). Given a planned forest road i , i.e., one finite linear segment which corresponds to the approximate axis of the proposed carriageway, its ecological risk $r_i \in [0,1]$ is given as follows:

$$r_i = H_i \cdot \frac{\sum_{j=1}^{m} \left(D_{ij} \cdot V_{ij} \right)}{m} \tag{3}$$

where H_i is the probability [0,1] of occurrence of a hazard factor along the road *i*. The hazard may be a combination of more than one hazard factor, where the individual probability values of hazard events should be combined to determine the probability of a specific sequence of multiple hazard events occurring. $D_{ij} \in [0,1]$ and $V_{ij} \in [0,1]$ are, respectively, the potential damage and the vulnerability of the m th ecosystem component receptor *j* for road *i*. These values derive from and summarise one or more specific indices, scaled with Equation (2) and selected depending on the characteristics and features of the area and the aspect considered. Weights (*w*) may be applied to indicators depending on the nature of each receptor and on users' knowledge and concern.

R (Equation (1)) equals the sum of r_i. Therefore, the cumulative probability *P* that the risk *R* of a plan may assume any value lower than or equal to the number of roads (*n*), that is, when all $r_i = 1$, is as follows:

$$P\,(R \leq n) = F\,(r) \qquad 0 \leq F(r) \leq 1 \tag{4}$$

If one arranges the observed values ri /n in ascending order, the roads to be rejected are those for which $\sum_{i=1}^{n} r_i/n$ = $F(r) \geq \omega$ where ω should be fixed not higher than a precautionary threshold of probability, in order to ensure that a significant effect is avoided. The value of ω is determined based on local conditions, the size of the plan and following the indications of biodiversity experts and authorities in charge of biodiversity conservation. In the event that two or more roads assume the same *r* value as *F*(*r*) approaches ω, the one with the lower priority amongst them should be rejected.

The above equations are conceptually close to the regional ecological risk (Landis and Wiegers 1997) for two reasons. First, when a source (the road) generates stressors (the hazards) that affect habitats important in the assessment end points (the components), the ecological risk will be high. Second, the application of the equations should incorporate real properties of ecological structures, the multitude of stressors and the geographical units of managers.

3.3.2.1 Hazard

First, a straightforward measure of the hazard level is needed. The hazard events that were considered in our case study are derived from logging activities. Although logging activities per se are not necessarily a hazard, especially in the study area, where silvicultural activities follow sustainable principles (i.e., close-to-nature silviculture) under strict law requirements and regulations, negative effects cannot be excluded a priori because many species and habitats could suffer from temporary activities and changes to forest characteristics. To estimate the hazard H_i derived from forest operations, the prescribed yield for the compartments, within which the proposed roads are located, was derived from forest working plans. This measure relates to the probability and frequency of forest logging, thus indicating the hazard. In this case, we used five classes of programmed yield, representing increasing probability of logging-related hazards H_i.

3.3.2.2 Value and vulnerability

To assess the value that indicates potential damage in ecological risk assessment and vulnerability of the ecosystem, three components were considered: habitats (*h*), plants (*p*) and animals (*a*). To obtain specific quantitative data, specific field surveys (Table 1) and existing databases (Table 2) were used. Given the wide variability of site and habitat conditions covered by the plan and its representativeness, the mean value of each index for each road was scaled with Equation (2), with x_{max} being the maximum observed value among the roads.

The centres of three concentric sampling plots (12, 25 and 50 m radii) and of two orthogonal transects (50 m long) were randomly selected through the random walk technique: the surveyor walked along the approximate axis of the road carriageway for 300 m and then positioned the centre of the plots according to a random azimuth and distance, within a maximum 50-m buffer distance from the road. This buffer distance was considered to be representative of the road edge: the area inside which the major direct disturbances may occur due to the establishment of a new road. In fact, this can be broken up into the regulatory width of 10 m of forest road, its verge and related adjacent slope, plus 30 m as a disturbance strip for tree vegetation (Trafela 1987) and 15 m as a precautionary strip (Figure 2). As a result of the random walk sampling, 30% of the surface within 50 m from the road axis was surveyed.

Several surveys were carried out following a field protocol, which is detailed in Table 1. An average time of 60 min was needed to survey a sampling unit. As it can be seen in this table, the variables were sampled inside different plot radii to account for their different spatial scale of variability. Top height (*St*), volume of fallen deadwood (*Cw*), volume of standing deadwood (*Sn*) and phytogeographic value (*Ph*) were used as indices of habitat value and they were then weighted using w = 0.7 for *St* and *Ph* and w = 0.3 for *Cw* and *Sn*. Therefore, habitat value Dh_i was as follows:

$$Dh_i = 0.7\left(\frac{St_i+Ph_i}{2}\right) + 0.3\left(\frac{Cw_i+Sn_i}{2}\right) \quad (5)$$

A hierarchical cluster analysis was carried out with abundance/dominance data for woody species: groups were created through the minimum variance method (Ward Jr. 1963). Stand-level forest typologies define meaningful assemblages of structural and compositional diversity and can be assumed as the basis for delineating relevant biodiversity units among the forest landscape, as well as distinctive silvicultural prescriptions (Barbati et al. 1999). The groups resulting from the cluster analysis were then referenced to a regional stand-level forest typology (Odasso 2002). Their quality (Phi) was assigned based on the stand-level forest-type biodiversity indices developed by Del Favero (2000). The vulnerability of habitat (Vh_i) to logging activities was assigned using the same stand-level forest typology.

The potential damage to the plant species was $Dp_i = (Ne_i + Nr_i)/2$ and took into account the scaled number of endemic species Ne_i and of the species listed in a provincial red list Nr_i (Prosser 2001). Species listed in Annex II of the Habitats Directive were absent. The computation of these species for each forest road was based on the information collected in the field (see Table 1) together with data derived from a GIS-based plant species atlas with a resolution of 900 m^2 (Prosser and Festi 2008). Plant species vulnerability Vp_i was the scaled richness of the plant species threatened by the construction and use of forest roads according to their biological traits.

Table 1: Variables collected in the field to calculate the habitat value and their explanation (DBH: diameter at breast height). Original values for each road are also reported.

Indicator name	**Plot shape and size**	**Measurement**	**Definition**	**Min-max (mean ± 95% conf. int.)**
Snags (*Sn*)	circle, 25-m radius	Volume (by height and DBH) of all standing dead trees (DBH > 10 cm, height > 2 m)	Indicator of naturalness and wilderness	0.3-27.4 (8.4±4.6) m^3ha^{-1}
Dominant height (*St*)	circle, 50-m radius	Mean height of the dominant trees	Surrogate index of the carrying capacity level of the stand (Susmel 1980)	16.5-32 (24.7±2.2) m
Logs (*Cw*)	2 perpendicular transects, each 50-m long	Volume of downed dead trees (DBH > 10 cm, length > 1.5 m), with the line intersect method (Marshall, Davis, and LeMay 2000)	Surrogate index of habitat quality for many living organisms (Harmon et al. 1988)	0-33.4 (7.6±4.4) m^3ha^{-1}
Phytogeographic value (*Ph*)	circle, 25-m radius	Assignment of each plot to one of the forest types which result from a cluster analysis of the canopy cover of all woody species. Each forest type has its phytogeographic value according to a regional standard forest typology (Odasso 2002)	Phytogeographic importance of the plant species which characterise each forest type (Del Favero 2000)	1-5 (2.7±0.4)

Table 2: Indicators derived from existing databases or expert judgment and their explanation. Original values for each road are reported.

Risk factor	**Indicator name**	**Measurement**	**Definition**	**Min-max (mean ± 95% conf. int.)**
Hazard	Hazard from forest operations (H_i)	Proportional to prescribed yield in the forest compartments where the road lie	Yield is proportional to the frequency and intensity of use of forest roads	10-75 (37±7) $m^3ha^{-1}y^{-1}$
Value	Endemic vascular species (Ne_i)	Mean number of endemic species in the quadrats of a Park's atlas that lie within a 50-m road buffer (Prosser and Festi 2004)	Indicator of biodiversity	0-25 (7.6±3.8)
	Red list vascular species (Nr_i)	Mean number of threatened plant species in the quadrats of a Park's atlas that lie within a 50-m	Indicator of biodiversity	0-8 (1.8±1.0)

Risk factor	Indicator name	Measurement	Definition	Min-max (mean ± 95% conf. int.)
		road buffer (Prosser and Festi 2004)		
	Animal species value (Da_i)	Summed score for the most relevant species which are present along the road	The species' score is proportional to its phenology, general rarity, chorology, reproductive success, regional rarity, habitat specialism, representativeness of the regional population and IUCN threat category	79-121 (95.7±6.4)
Vulnerability	Habitat vulnerability (Vh_i)	Mean vulnerability score of each forest type weighted by the number of sampling plots belonging to each forest type	Vulnerability to silviculture based on the possible influence of treatments on natural dynamics	1-5 (2.3±0.5)
	Plant species vulnerability (Vp_i)	Scaled number of threatened plant species in the quadrats of a Park's atlas that lie within a 50-m road buffer (Prosser and Festi 2004)	Include the species threatened by forest roads construction and use according to expert judgment	0-100 (43.7±15.6)
	Animal species vulnerability (Vk_i)	Summed vulnerability score for the most relevant animal species which are present along the road	Expert judgment of vulnerability along the road in terms of local sensitivity of each population to forest road construction and use	28-43 (35.3±2.4)
	Brown bear and capercaillie vulnerability (Vut_i)	Mean vulnerability score of brown bear and capercaillie	Based on a detailed analysis of their sightings and distribution maps	1-3 (1.9±0.28)

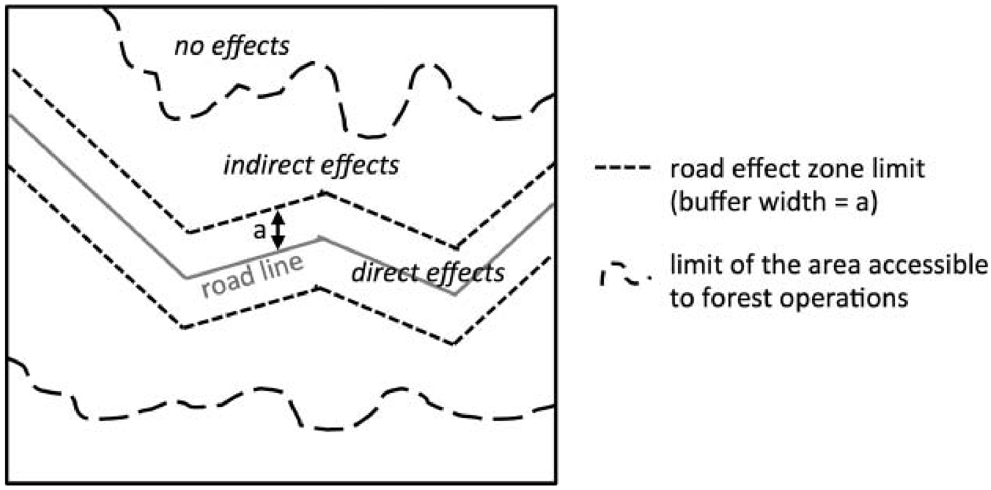

Figure 2: Schematic representation of the distinction between direct effects of a forest road which extend within a road effect zone limit and indirect effects which depend on the area accessible to forest operations. In the case study presented here, a D 50 m and the width of area accessible to forest operations depends on terrain slope (maximum suitable for forest operations D 120%), time available for a round trip (30 min), and the mean speed of forest workers (3.7 km h^{-1} on flat terrain and 350 m h^{-1} of elevation gain between 10% and 120% slope).

The potential damage to the animal species Da_i is given by the scaled sum of the values for all the relevant animal species that are present along the road, representing a synthetic value derived from several criteria (see also Table 2). The vulnerability Vk_i of the most relevant animal species was derived from their sensitivity and from the expected habitat loss and/or degradation. The vulnerability Vut_i of two umbrella species in the area, brown bear (*Ursus arctos* L.) and capercaillie (*Tetrao urogallus* L.), was reported separately from the other animal species. A more detailed analysis was carried out for these two species because they are umbrella species and they are of particular conservation concern in the Italian Alps (Suter, Graf, and Hess 2002; Roberge and Angelstam 2004). Finally, animal species vulnerability was $Va_i = (Vk_i + Vut_i)/2$.

3.3.3 Indirect effects related to accessibility

To assess the indirect and large-scale effects of forest roads connected to forest operations, we used a forest accessibility model. A spatial analysis was performed following the approach of the cost-distance analysis to define the forest accessibility in terms of the maximum accessible distance from a forest road within a certain walking time. This method enables us to quantify the area on which forest operations may occur following the creation of new road segments. Furthermore, it identifies areas where relevant mitigation actions should be applied.

The analysis is based on the travel time of an operator walking into a forest environment. First, the duration of a round trip is set as the maximum acceptable for an operator during daily work. Therefore, the furthest point from the road reached by the operator will depend on this duration. This distance is calculated by considering the terrain grade as a correcting factor of the planar distance in a least cost path analysis based on the relationships between slope and walking speed, as proposed by Tobler (1993) and implemented in GIS environment (e.g., Pettebone, Newman, and Theobald 2009; Ciesa, Grigolato, and Cavalli 2014; Doherty et al. 2014). A GIS routine determines the least cost path from any point on the road, which is the cheapest route relative to the cost unit defined by a cost raster of cell crossing times within the maximum time available for a round trip.

The cost value is obtained by multiplying the percentage slope α from a digital terrain model (DTM) by a friction factor *k*:

$$k = \frac{t_{\alpha>q}}{t_{\alpha\leq q}} = \frac{\Delta q}{v_{\alpha>q}} / \frac{l}{v_{\alpha\leq q}} \quad (6)$$

where $t_{\alpha\leq q}$ and $t_{\alpha>q}$ are the amounts of time in seconds needed to cross a raster cell below a certain slope threshold (q) and to gain 1 m in elevation above this threshold, respectively. $\Delta q = l\,(\alpha/100)$ is the elevation gain in meters and $v_{\alpha>q}$ is the speed expressed in altitudinal gain in meters per second above the threshold *q*, *l* is the average length expressed in meters between the side and the diagonal of the DTM cells, while $v_{\alpha\leq q}$ is the speed in meters per second below the slope threshold *q*.

The method, as proposed by Hippoliti (1976), for forest workers in mountainous areas, assumes that an operator walks at a fixed speed under a certain threshold of slope, above which the time needed to gain elevation increases proportionally to the slope. The area that is made accessible to forest operations becomes a measure of potentially adverse indirect impacts. This approach can also be used to compare different alternative scenarios or plan revisions, by comparing the size and the distribution of the area that is made accessible.

3.3.3.1 Improvement of forest accessibility by the road plan

The forest accessibility analysis was applied to define areas of indirect effect and specifically to compare proposals A, B and possible alternative solutions. The friction *k* (Equation (6)) was calculated using q = 10% and $v_{\alpha\leq q}$ = 10.3 m s^{-1} (3.7 km h^{-1}). Then, on terrains with slope higher than 10% and up to 120% a walking speed ($v_{\alpha>q}$) of 0.097 m in altitudinal gain per second (350 m h^{-1}) was used (Hippoliti 1976). We defined 30 min as the time limit for a round trip from each forest road segment, a value considered to be the maximum acceptable for an 8-h working day of a forest operator (Hippoliti 1976; Grigolato, Pellegrini, and Cavalli 2013). The forest accessibility analysis was based on a DTM with a raster resolution of 2 m. The GIS analysis was carried out using the software ESRI ArcGIS® 10.1. An analysis of the area made accessible was used to evaluate possible alternative road scenarios and to identify relevant mitigation actions.

3.4 Results

3.4.1 Ecological risk

Following a precautionary approach, we chose to consider the longest *lb* alternative for the road *l* because option *la* was likely to increase the risk for *Accipitridae* species, as it was partly formed by a cableway. Finally, only the road *n* proposed in 2012 was rejected by considering α = 0.3 (Figure 3(a)). Road *n* resulted in having, in particular, a high hazard probability, high values for deadwood and red-listed species and an overall high vulnerability for all the habitats and species considered.

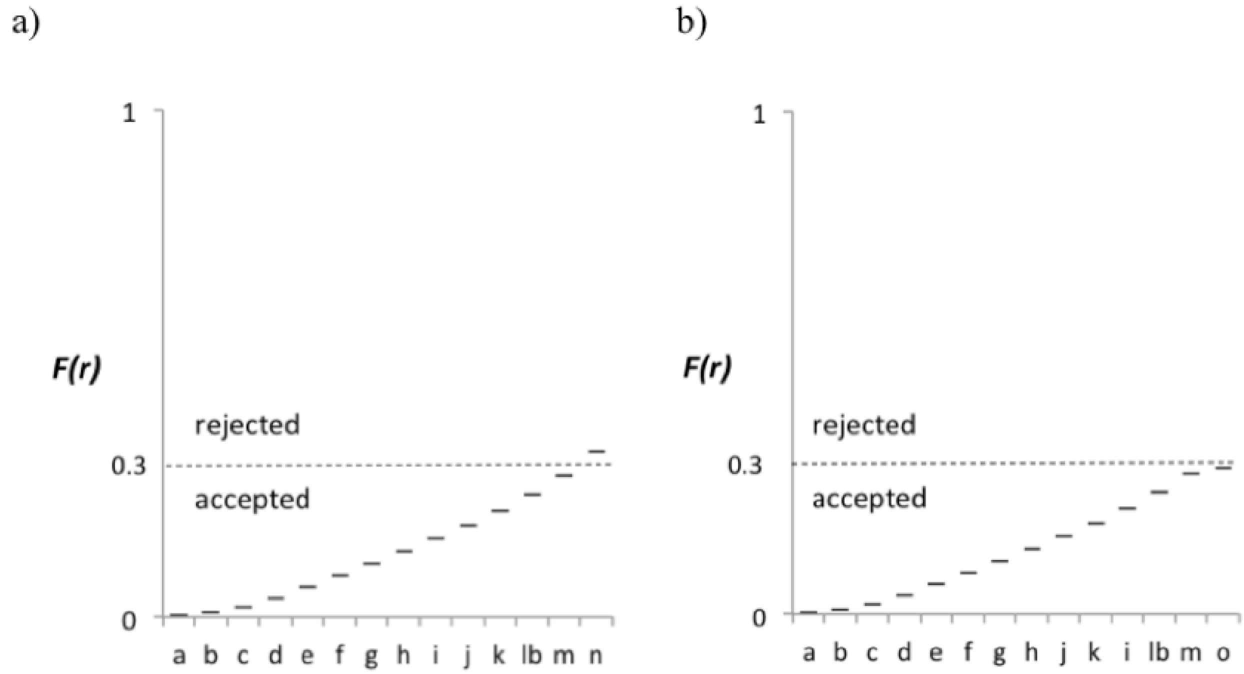

Figure 3. The roads (*a*–*o*) proposed in the 2012 revision of the forest network plan and the cumulative probability *F*(*r*) of R D 1 (a). Based on a threshold of acceptable risk (ϖ) = 0.3, the road n in the plan was rejected, and an alternative route *o* was proposed (b).

3.4.2 Forest accessibility

The accessible forest area in 2006 was 3,467 ha to which 269 ha would have been added by the forest road plan approved in 2008. After acceptance of the alternative o, a total of 314 ha would have been made accessible by the plan of 2012 compared to 2006. Therefore, the 2012 plan revision would have made accessible and exploitable an additional 45 ha (or 55 ha if the alternative route *lb* for the road *l* is considered) compared to the plan of 2008.

The magnitude of the impacts depends on the conservation value of the area made accessible. We suggest that old-growth features, in particular standing dead trees, are indicators of a high probability that forest operations will result in a reduction of habitat conservation status (Angelstam et al. 2003; Harmon et al. 1988; Cantarello and Newton 2008; Verkerk et al. 2011). A study conducted in a nearby region by Sitzia et al. (2012) showed that silver fir–beech forests accumulate an average standing deadwood of 14.1 ± 8.5 m^3ha^{-1} after 50 years of management cessation, while those managed with low-intensity shelterwood systems had an average standing deadwood of 1.7 ± 3.4 m^3ha^{-1}. In our case study, we sampled standing deadwood in a total of 63 plots inside the area made potentially accessible by the planned forest roads, which was 314 ha. Of those plots, four had values of standing deadwood higher than measured by Sitzia et al. (2012), that is, 6% of the plots, which expressed in terms of new accessible forest surface (314 ha) correspond to 20 ha.

3.4.3 Identification of alternatives

As a consequence of what resulted from the analysis of risk and forest accessibility, we deemed that the possibility of an alternative solution for the road *n* was needed because we found high biodiversity value and vulnerability along this road. Moreover, with the advice of the Park administration, an alternative route *o* was identified in order to avoid areas with higher value and vulnerability and reduce the total cumulative risk (see Figures 1 (f) and 3 (b)).

3.4.4 Mitigations and restrictions to accessibility

The mitigation actions included the setting aside of 20 ha of the area accessible to forest operations where woodlands with high standing deadwood volume could be heavily disturbed by the use of the new roads. This calculation derived from coupling the information gathered on deadwood features with the areas made accessible by the forest road plan. Additionally, a set of more general mitigation actions was prescribed: all related projects must undergo an AA, activities must not be carried out during the reproductive season of the most sensitive species and the time of disturbance must be reduced to a minimum. Attention must be given to residual cumulative effects by avoiding the simultaneous construction of more than one road during a given period. All roads must be open only to logging traffic and other specifically authorised uses.

3.5 Discussion

Article 6(3) of the Habitats Directive does not include specific information on the content of the AA. The Directive only indicates that the assessment must be “appropriate” and the relevant public authority must ascertain the absence of adverse effects by the plan or the project on the site’s integrity (Haumont 2015). For ECJ, this implies that the verification of the absence of significant effects must be carried out by applying the best scientific knowledge (ECJ 2004) and by providing complete, precise and definitive findings and conclusions (ECJ 2007). No reasonable scientific doubt should remain as to the absence of such effects (ECJ 2004).

AA is carried out applying several different methods (Söderman 2009; Therivel 2009). Nevertheless, an indication on “how to” assess impacts is still needed and the proposed method helps in casting light on a number of recognised AA shortcomings: low quality of the assessments, poor knowledge base and lack of cumulative effects assessment (Sundseth and Roth 2013). Similar limitations were highlighted for Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA). For example, the assessment of impacts on biodiversity in EIA usually lacks evidence-based assessment techniques, a delimitation of study areas on an ecological basis, measurable indicators and quantitative predictions (e.g., Geneletti 2002; Geneletti 2006; Gontier, Balfors, and

Mörtberg 2006). Indeed, integrating the identification of ecological risk and the delimitation of the area under investigation could also benefit the assessment of biodiversity impacts in these other two assessment types.

Currently, roads represent an important pressure and threat to habitats and species of Community interest. This activity is reported among the most high-ranked pressures and threats for species associated with woodland and forest ecosystems (European Environment Agency 2015). The method presented here helps to deal with the pressures and threats to the ecosystem, as it combines quantitative and qualitative analyses with data gathered through field surveys in areas potentially linked to the construction of new roads. This combination can be used to relate ecological processes to wider contexts, helps to reduce uncertainty and enables the inclusion of long-term effects in impact prediction (Karlson, Mörtberg, and Balfors 2014). The significance of effects is generally understood to be context specific (Bryan 2012). The assessment of ecological risk and the accessibility model are appropriate for local specific information. The method takes into account local environmental features (e.g., habitats, plants and wildlife) and specific threats of the area under investigation (e.g., forest exploitation). Furthermore, the replicability of the method is ensured by the possibility of choosing different indices to be entered in the calculation of the ecological risk. Indices vary on the basis of whether a plan or project is considered, which threats are brought by the activity and which are the conservation objectives of the area and the biodiversity features likely to be impacted. In agreement with the opinion of the ECJ (Kokott 2004, I-7427, paragraph 73), we attached "greater weight […] to doubts as to the absence of irreversible effects or effects on particularly rare habitats or species than to doubts as to the absence of reversible or temporary effects or the absence of effects on relatively common species or habitats". For example, in this case study, deadwood was one of the indicators used, as it represents an ecological niche for several taxa in nearby silver fir-beech forests like certain lichen (Nascimbene, Dainese, and Sitzia 2013) and beetle species (Sitzia et al. 2015). In other AAs, different indicators will suit the particular cases. The selection of indicators and indices should consider their link to the conservation status of the habitats and species analysed and should be related to the conservation objectives of the Natura 2000 sites, a choice that complies with regulatory requirements. For example, deadwood features are used as indicators in different member states to properly fulfil the requirements of a good conservation status of forest habitats (European Commission 2015b). In this view, the possibility of selecting indicators is important, as member states usually apply different approaches to assess the conservation status of habitats and species (Opdam, Broekmeyer, and Kistenkas 2009). Future work should, therefore, test the method in other types of plans and projects, regulatory frameworks and land uses other than forests.

The method enabled us to test alternative solutions of the plan and detect mitigation measures. To achieve this, we capitalised on the data made available by field surveys and from the use of spatially explicit characteristics of the accessibility model. Moreover, the accessibility model for forest workers can be also used to model the area made accessible to hikers after road construction, which is another potential source of disturbance (Wolf, Hagenloh, and Crof 2012). However, like in any other impact assessment study, the final decision of mitigation actions, weights and alternatives is a part of the assessment which should involve stakeholder participation (Wathern 1988) and cannot be part of a standard methodology.

The ECJ states that the conclusion of the appropriate assessment "is, of necessity, subjective in nature" (Kokott 2004, I-7435, paragraph 107). Also, the present method incorporates a subjective choice of the relevant indicators, to have a view of the whole environment possibly affected by the plan. However, it obliges us to explicitly declare the factors used in the impact assessment and their interactions, reducing ambiguity (Opdam, Broekmeyer, and Kistenkas 2009). An important step in the method's application is the choice of the acceptable level of risk (ϖ related to Equation (4)). This point does not contradict the precautionary principle included in Article 174(2) European Community Treaty because this principle should be applied proportionally to the assumed risk. Defining what can be considered to be an 'acceptable' level of risk for society is highly a political responsibility (European Commission 2000b; Kokott 2004; Opdam, Broekmeyer, and Kistenkas 2009). Attention must be given to the probability of occurrence of harmful events and to their extent and typology, even though "in many areas there is considerable scientific uncertainty as to cause and effect" (Kokott 2004, I-7432, paragraph 97). The threshold ϖ determines the point at which the precautionary principle is triggered (Stokes 2005). Therefore, the interpretation of the effects is expressed in probabilistic terms by predicting a level of risk rather than predicting the dimension of change (Opdam, Broekmeyer, and Kistenkas 2009), which is not a common practice in EIAs.

The application of this method is reliable for areas where data on species distribution and specific habitat features are available and scientifically solid. Habitat suitability models, in particular, may further assist in the application of our method.

3.6 Conclusion

We showed that the method allows summarising a set of specific risks at ecosystem and population levels related to local environmental conditions and to the Natura 2000 sites' conservation objectives. This application of the ecological risk extends that of Andersen, Thompson and Boykin (2004) by coupling relative risk assessment with a model of forest accessibility which permits us to quantify the cumulative risk of road construction and to quantitatively compare the effects of forest plan variants. The method fits with the requirements of the Natura 2000 regulations because it includes the perceived value of the affected environment, the magnitude, the spatial extent and the duration of anticipated change, the habitat and species vulnerability and the cumulative impacts of the construction of roads.

Even though AA has the potential to inform on the negative effects of plans and projects, we emphasise the need for novel approaches in spatial planning that are able to couple human activities with the ecosystem (Vikolainen, Bressers, and Lulofs 2013). Moreover, planning should incorporate different approaches enabling the inclusion of the variety of ways in which people value nature, thus avoiding the dominance of one type of value, i.e., economic, over others, i.e., non-economic (Ferranti et al. 2014).

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3.7 References

Andersen, M.C., B. Thompson, and K. Boykin. 2004. "Spatial Risk Assessment Across Large Landscapes with Varied Land Use: Lessons from a Conservation Assessment of Military Lands." *Risk Analysis* 24 (5): 1231–1242. doi: 10.1111/j.0272-4332.2004.00521.x.

Angelstam, P.K., R. Bütler, M. Lazdinis, G. Mikusi, and J.-M. Roberge. 2003. "Habitat Thresholds for Focal Species at Multiple Scales and Forest Biodiversity Conservation — Dead Wood as an Example." *Annales Zoologici Fennici* 40 (6): 473–482.

Avon, C., L. Bergès, Y. Dumas, and J.-L. Dupouey. 2010. "Does the Effect of Forest Roads Extend a Few Meters or More into the Adjacent Forest? A Study on Understory Plant Diversity in Managed Oak Stands." *Forest Ecology and Management* 259 (8): 1546–1555. doi: 10.1016/j. foreco.2010.01.031.

Barbati, A., P. Corona, G. Carraro, R. Del Favero, M. Dissegna, C. Lasen, and M. Marchetti 1999. "Developing Biodiversity Assessment on a Stand Forest Type Management Level in North-Eastern Italy." *Annali Accademia Italiana di Scienze Forestali* 48: 157–176.

Benítez-López, A., R. Alkemade, and P.A. Verweij. 2010. "The Impacts of Roads and Other Infrastructure on Mammal and Bird Populations: A Meta-analysis." *Biological Conservation* 143: 1307–1316. doi: 10.1016/j.biocon.2010.02.009.

Beunen, R. 2006. "European Nature Conservation Legislation and Spatial Planning: For Better or for Worse." *Journal of Environmental Planning and Management* 49 (4): 605–619. doi: 10.1080/09640560600747547.

Beunen, R., W.G.M. van der Knaap, and G.R. Biesbroek. 2009. "Implementation and Integration of EU Environmental Directives. Experiences from the Netherlands." *Environmental Policy and Governance* 19 (1): 57–69. doi:10.1002/eet.495.

BMVI. "Methodological Guideline for Impact Assessment of Transportation Infrastructure Significantly Affecting Natura 2000 Sites (Guideline for IA). Guidance on the Provisions of Article 6(3,4) of the Habitats Directive." Berlin: Bundesministerium für Verkehr, Bau- und Wohnungswesen. Accessed November 10, 2015. http://www.kifl.de/english/guidance-and-training.htm

Bryan, S. 2012. "Contested Boundaries, Contested Places: The Natura 2000 Network in Ireland." *Journal of Rural Studies* 28 (1): 80–94. doi: 10.1016/j.jrurstud.2011.09.002.

Cantarello, E., and A.C. Newton. 2008. "Identifying Cost-Effective Indicators to Assess the Conservation Status of Forested Habitats in Natura 2000 Sites." *Forest Ecology and Management* 256 (4): 815–826. doi: 10.1016/j.foreco.2008.05.031.

Cavalli, R., and S. Grigolato. 2009. "Influence of Characteristics and Extension of a Forest Road Network on the Supply Cost of Forest Woodchips." *Journal of Forest Research* 15 (3): 202–209. doi: 10.1007/s10310-009-0170-4.

Ciesa, M., S. Grigolato, and R. Cavalli. 2014. “Analysis on Vehicle and Walking Speeds of Search and Rescue Ground Crews in Mountainous Areas.” *Journal of Outdoor Recreation and Tourism* 5-6: 48–57. doi: 10.1016/j.jort.2014.03.004.

Coffin, A.W. 2007. “From Roadkill to Road Ecology: A Review of the Ecological Effects of Roads.” *Journal of Transport Geography* 15(5): 396–406. doi: 10.1016/j.jtrangeo.2006.11.006.

Dana, G.V., A.R. Kapuscinski, and J.S. Donaldson. 2012. “Integrating Diverse Scientific and Practitioner Knowledge in Ecological Risk Analysis: A Case Study of Biodiversity Risk Assessment in South Africa.” *Journal of Environmental Management* 98: 134–146.

Del Favero, R., ed. 2000. *Biodiversità e indicatori nei tipi forestali del Veneto*. Venezia Mestre: Direzione Foreste ed Economia Montana della Regione Veneto.

Doherty, P.J., Q. Guo, J. Doke, and D. Ferguson. 2014. “An Analysis of Probability of Area Techniques for Missing Persons in Yosemite National Park.” *Applied Geography* 47: 99–110. doi: 10.1016/j.apgeog.2013.11.001.

ECJ. 2004. *Case C-127/02 Landelijke Vereniging tot Behoud van de Waddenzee, Nederlandse Vereniging tot Bescherming van Vogels vs Staatssecretaris van Landbouw, Natuurbeheer en Visserij*. Judgment of the Court (Grand Chamber) 7 September 2004. European Court Reports 2004 I-07448. Luxembourg: European Court of Justice. Accessed November 14, 2015. http://curia.europa.eu/juris/liste.jsf?language=en&num=C-127/02

ECJ. 2007. Case C-304/05 *Commission of the European Communities vs. Italian Republic*. Judgement of the Court (Fourth Chamber) 20 September 2007. European Court Reports 2007 I- 07495. Luxembourg: European Court of Justice. Accessed November 15, 2015. http://curia.europa.eu/juris/liste.jsf?language=en&num=C-304/05

European Commission. 2000a. *Managing Natura 2000 Sites: The Provisions of Article 6 of the Habitats Directive 92/43/EEC*. Luxembourg: Office for Official Publications of the European Communities.

European Commission. 2000b. *Communication from the Commission on the Precautionary Principle, COM(2000) 1 Final*. Brussels: European Commission.

European Commission. 2002. *Assessment of Plans and Projects Significantly Affecting Natura 2000 Sites. Methodological Guidance on the Provisions of Article 6(3) and (4) of the Habitats Directive 92/43/EEC*. Vol. 6. Luxembourg: Office for Official Publications of the European Communities.

European Commission. 2010. *Guidance Document. Non-Energy Mineral Extraction and Natura 2000*. Luxembourg: Publications Office of the European Union. doi: 10.2779/98870

European Commission. 2011a. *Guidance Document. Wind Energy Developments and Natura 2000*. Luxembourg: Publications Office of the European Union. doi:

10.2779/98894

European Commission. 2011b. *Guidance Document. The Implementation of the Birds and Habitats Directives in Estuaries and Coastal Zones with Particular Attention to Port Development and Dredging*. Luxembourg: Publications Office of the European Union.

European Commission. 2012a. *Guidance Document on Article 6(4) of the "Habitats Directive" 92/ 43/EEC*. Vol. 6. Luxembourg: Office for Official Publications of the European Communities.

European Commission. 2012b. *Guidance Document on Inland Waterway Transport and Natura 2000. Sustainable Inland Waterway Development and Management in the Context of the EU Birds and Habitat Directives*. Luxembourg: European Commission.

European Commission. 2012c. *Guidance on Aquaculture and Natura 2000. Sustainable Aquaculture in the Context of Natura 2000 Network*. Luxembourg: European Commission.

European Commission. 2015a. *Natura 2000 and Forests. Part I–II, Technical Report – 2015–088*. Luxembourg: Office for Official Publications of the European Communities.

European Commission. 2015b. *Natura 2000 and Forests. Part III – Case Studies, Technical Report– 2015–089*. Luxembourg: Office for Official Publications of the European Communities.

European Environment Agency. 2010a. "Brenta. Factsheet Filled with Data from Natura 2000 Data Set." Luxembourg: European Environment Agency. http://eunis.eea.europa.eu/sites/IT3120159/general

European Environment Agency. 2010b "Dolomiti Di Brenta. Factsheet Filled with Data from Natura 2000 Data Set." Luxembourg: European Environment Agency. http://eunis.eea.europa.eu/sites/IT3120177/general.

European Environment Agency. 2015. *State of Nature in the EU. Results from Reporting Under the Nature Directives 2007–2012*. Vol. 2, EEA Technical Report. Luxembourg: European Environment Agency.

Ferranti, F., R. Beunen, and M. Speranza. 2010. "Natura 2000 Network: A Comparison of the Italian and Dutch Implementation Experiences." *Journal of Environmental Policy and Planning* 12 (3): 293–314. doi: 10.1080/1523908X.2010.505417.

Ferranti, F., E. Turnhout, R. Beunen, and J.H. Behagel. 2014. "Shifting Nature Conservation Approaches in Natura 2000 and the Implications for the Roles of Stakeholders." *Journal of Environmental Planning and Management* 57 (11): 1642–1657. doi: 10.1080/09640568.2013.827107.

Festi, F. and F. Prosser. 2008. *Flora del Parco Naturale Adamello Brenta*. Rovereto, Italy: Edizioni Osiride.

Fock, H.O., M. Kloppman, and V Stelzenmüller. 2011. "Linking Marine Fisheries to

Environmental Objectives: A Case Study on Seafloor Integrity Under European Maritime Policies." *Environmental Science and Policy* 14: 289–300. doi: 10.1016/j.envsci.2010.11.005.

Forman, R.T.T. 2000. "Estimate of the Area Affected Ecologically by the Road System in the United States." *Conservation Biology* 14 (1): 31–35.

Forman, R.T.T., D.S. Friedman, D. Fitzhenry, J.D. Martin, A.S. Chen, and L.E. Alexander. 1997. "Ecological Effects of Roads: Toward Three Summary Indices and an Overview for North America." In *Habitat Fragmentation and Infrastructure*, edited by K. Canters, 40–54. Delft: Ministry of Transport, Public Works and Water Management.

Forman, R.T.T., D. Sperling, J.A. Bissonette, A.P. Clevenger, C.D. Cutshall, V.H. Dale, L. Fahrig, et al. 2003. *Road Ecology: Science and Solutions*. Washington, DC: Island Press.

Geneletti, D. 2006. "Some Common Shortcomings in the Treatment of Impacts of Linear Infrastructures on Natural Habitat." *Environmental Impact Assessment Review* 26 (3): 257–267.

Geneletti, D. 2002. *Ecological Evaluation for Environmental Impact Assessment*. Utrecht: Geographical Studies.

Ghaffariyan, M.R., K. Stampfer, and J. Sessions. 2010. "Optimal Road Spacing of Cable Yarding Using a Tower Yarder in Southern Austria." *European Journal of Forest Research* 129 (3): 409–416. doi: 10.1007/s10342-009-0346-7.

Gontier, M., B. Balfors, and U. Mörtberg. 2006. "Biodiversity in Environmental Assessment –Current Practice and Tools for Prediction." *Environmental Impact Assessment Review* 26 (3): 268–286.

Grigolato, S., M. Pellegrini, and R. Cavalli. 2013. "Temporal Analysis of the Traffic Loads on Forest Road Networks." *iForest* 6 (4): 255–261. doi: 10.3832/ifor0773-006.

Gucinski, H., M.J. Furniss, R.R. Ziemer, and M.H. Brookes. 2001. *Forest Roads: A Synthesis of Scientific Information*. Washington, DC: United States Department of Agriculture Forest Service.

Gumus, S., H.H. Acar, and D. Toksoy. 2008. "Functional Forest Road Network Planning by Consideration of Environmental Impact Assessment for Wood Harvesting." *Environmental Monitoring and Assessment* 142 (1–3): 109–116. doi: 10.1007/s10661-007-9912-y.

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, et al. 1988. "Ecology of Coarse Woody Debris in Temperate Ecosystems." *Advances in Ecological Research* 15: 133–276.

Haskell, D.G. 2000. "Effects of Forest Roads on Macroinvertebrate Soil Fauna of the Southern Appalachian Mountains." *Conservation Biology* 14 (1): 57–63.

Haumont, F. 2015. "Appropriate Impact Assessment. The Key to Effective Integration of Nature Conservation Issues into Land-Use Planning." In *The Habitats Directive in its*

EU Environmental Law Context: European Nature's Best Hope? edited by C.-H. Born, A. Cliquet, H. Schoukens, D. Misonne and G. Van Hoorick, 93–100. New York: Routledge.

Hayati, E., B. Majnounian, and E. Abdi. 2012. "Qualitative Evaluation and Optimization of Forest Road Network to Minimize Total Costs and Environmental Impacts." *iForest* 5 (3): 121–125. doi: 10.3832/ifor0610-009.

Hippoliti, G. 1976. "Sulla Determinazione Delle Caratteristiche Della Rete Viabile Forestale." *L'Italia Forestale e Montana* 31: 241–255.

Karlson, M., and U. Mörtberg. 2015. "A Spatial Ecological Assessment of Fragmentation and Disturbance Effects of the Swedish Road Network." *Landscape and Urban Planning* 134: 53–65. doi: 10.1016/j.landurbplan.2014.10.009.

Karlson, M., U. Mörtberg, and B. Balfors. 2014. "Road Ecology in Environmental Impact Assessment." *Environmental Impact Assessment Review* 48: 10–19. doi: 10.1016/j.eiar.2014.04.002.

Kati, V., T. Hovardas, M. Dieterich, P.L. Ibisch, B. Mihok, and N. Selva. 2015. "The Challenge of Implementing the European Network of Protected Areas Natura 2000." *Conservation Biology* 29 (1): 260–270. doi: 10.1111/cobi.12366.

Kokott, J. 2004. *Case C-127/02 Opinion of Advocate General Kokott Delivered on 29 January 2004*. European Court Reports 2004 I-07409. Luxembourg: European Court of Justice. Accessed November 15, 2015. http://curia.europa.eu/juris/celex.jsf?celex=62002CC0127&lang1=en&lang2=DE&type=TXT&ancre=.

Lackey, R.T. 1998. "Fisheries Management: Integrating Societal Preference, Decision Analysis, and Ecological Risk Assessment." *Environmental Science and Policy* 1: 329–335.

Landis, W.G., and J.A. Wiegers. 1997. "Design Considerations and a Suggested Approach for Regional and Comparative Ecological Risk Assessment." *Human and Ecological Risk Assessment* 3 (3): 287–297.

Lawrence, D.P. 2007a. "Impact Significance Determination – Designing an Approach." *Environmental Impact Assessment Review* 27 (8): 730–754. doi: 10.1016/j.eiar.2007.02.012.

Lawrence, D.P. 2007b. "Impact Significance Determination – Pushing the Boundaries." *Environmental Impact Assessment Review* 27 (8): 770–788. doi: 10.1016/j.eiar.2007.02.010.

Legendre, P., and L. Legendre. 1998. *Numerical Ecology*. Vol. 2. Amsterdam: Elsevier Science.

Lugo, A.E., and H. Gucinski. 2000. "Function, Effects, and Management of Forest Roads." *Forest Ecology and Management* 133 (3): 249–262.

Marshall, P.L., G. Davis, and M.V. LeMay 2000. *Using Line Intersect Sampling for Coarse Woody Debris. Technical Report TR-003*. Vancouver: Research Section, Vancouver

Forest Region, BCMOF.

Nascimbene, J., M. Dainese, and T. Sitzia. 2013. “Contrasting Responses of Epiphytic and Dead Wood-Dwelling Lichen Diversity to Forest Management Abandonment in Silver Fir Mature Woodlands.” *Forest Ecology and Management* 289: 325-332.

Odasso, M. 2002. *I tipi forestali del Trentino: catalogo, guida al riconoscimento, localizzazione e caratteristiche ecologico-vegetazionali*. Viote Monte Bondone, Trento: Centro di Ecologia Alpina.

Opdam, P.F.M., M.E.A. Broekmeyer, and F.H. Kistenkas. 2009. “Identifying Uncertainties in Judging the Significance of Human Impacts on Natura 2000 Sites.” *Environmental Science and Policy* 12 (7): 912–921. doi: 10.1016/j.envsci.2009.04.006.

Pentek, T., D. Pičman, I. Potočnik, P. Dvorščak, and H. Nevečerel. 2005. “Analysis of an Existing Forest Road Network.” *Croatian Journal of Forest Engineering* 26 (1): 39–50.

Pettebone, D., P. Newman, and D. Theobald. 2009. “A Comparison of Sampling Designs for Monitoring Recreational Trail Impacts in Rocky Mountain National Park.” *Environmental Management* 43 (3): 523–532. doi: 10.1007/s00267-008-9261-9.

Prosser, F. 2001. *Lista rossa della flora del Trentino: pteridofite e fanerogame*. Rovereto, Italy: Osiride.

Ries, L., R.J. Fletcher, J. Battin, and T.D. Sisk. 2004. “Ecological Responses to Habitat Edges: Mechanisms, Models, and Variability Explained.” *Annual Review of Ecology, Evolution, and Systematics* 35 (1): 491–522. doi: 10.1146/annurev.ecolsys.35.112202.130148.

Roberge, J.M., and P.E.R. Angelstam. 2004. “Usefulness of the Umbrella Species Concept as a Conservation Tool.” *Conservation Biology* 18 (1): 76–85.

Robinson, C., P.N. Duinker, and K.F. Beazley. 2010. “A Conceptual Framework for Understanding, Assessing, and Mitigating Ecological Effects of Forest Roads.” *Environmental Reviews* 18: 61–86. doi: 10.1139/A10-002.

Scolozzi, R., and D. Geneletti. 2011. “Spatial Rule-Based Assessment of Habitat Potential to Predict Impact of Land Use Changes on Biodiversity at Municipal Scale.” *Environmental Management* 47 (3): 368–383.

Scott Wilson, Levett-Therivel Sustainability Consultants, Treweek Environmental Consultants, and Land Use Consultants. 2006. *Appropriate Assessment of Plans*. Scott Wilson, Levett-Therivel Sustainability Consultants, Treweek Environmental Consultants, Land Use Consultants, August 2006. http://www.sea-info.net/files/general/Appropriate_Assessment_of_Plans.pdf.

Sitzia, T., T. Campagnaro, E. Gatti, M. Sommacal, and D.J. Kotze. 2015. “Wildlife Conservation Through Forestry Abandonment: Responses of Beetle Communities to Habitat Change in the Eastern Alps.” *European Journal of Forest Research* 134(3): 511–524. doi: 10.1007/s10342-015-0868-0

Sitzia, T., and G. Trentanovi. 2011. "Maggengo Meadow Patches Enclosed by Forests in the Italian Alps: Evidence of Landscape Legacy on Plant Diversity." *Biodiversity and Conservation* 20 (5): 945–961. doi: 10.1007/s10531-011-0006-3.

Sitzia, T., G. Trentanovi, M. Dainese, G. Gobbo, E. Lingua, and M. Sommacal. 2012. "Stand Structure and Plant Species Diversity in Managed and Abandoned Silver Fir Mature Woodlands." *Forest Ecology and Management* 270: 232–238. doi: 10.1016/j.foreco.2012.01.032.

Söderman, T. 2009. "Natura 2000 Appropriate Assessment: Shortcomings and Improvements in Finnish Practice." *Environmental Impact Assessment Review* 29 (2): 79–86. doi: 10.1016/j.eiar.2008.04.001.

Spellerberg, I.F. 1998. "Ecological Effects of Roads and Traffic: A Literature Review." *Global Ecology and Biogeography Letters* 7: 317–333.

Stokes, E.R. 2005. "Liberalising the Threshold of Precaution – Cockle Fishing, the Habitats Directive, and Evidence of a New Understanding of 'Scientific uncertainty'." *Environmental Law Review* 7 (3): 206–214. doi: 10.1350/enlr.2005.7.3.206.

Sundseth, K., and I. Roth. 2013. *Study on Evaluating and Improving Permitting Procedures Related to Natura 2000 Requirements Under Article 6.3 of the Habitats Directive 92/43/EEC*. Final Report. Contract No. 07.0307/2012/623211/SER/B3. Ecosystems Ltd.

Susmel, L. 1980. *Normalizzazione delle foreste alpine*. Padova: Liviana Editrice S.p.A.

Suter, G.W. 1993. *Ecological Risk Assessment*. Boca Raton, FL: Lewis Publishers.

Suter, G.W., L.W. Barnthouse, and R.V. O'Neill. 1987. "Treatment of Risk in Environmental Impact Assessment." *Environmental Management* 11 (3): 295–303.

Suter, G.W., R.F. Graf, and R. Hess. 2002. "Capercaillie (*Tetrao urogallus*) and Avian Biodiversity: Testing the Umbrella–Species Concept." *Conservation Biology* 16 (3): 778–788. doi: 10.1046/j.1523-1739.2002.01129.x.

Therivel, R. 2009. "Appropriate Assessment of Plans in England." *Environmental Impact Assessment Review* 29 (4): 261–272. doi: 10.1016/j.eiar.2009.01.001.

Tobler, W. 1993. *Three Presentations on Geographical Analysis and Modeling*. NCGIA Technical Report 93-1. Santa Barbara, California: National Center for Geographic Information and Analysis. http://escholarship.org/uc/item/05r820mz.

Trafela, E. 1987. "The Influence of the Construction of Forest Roads on Forest Production." *Zbornik Gozdarstva in Lesarstva* 29: 85–140.

Trombulak, S.C., and C.A. Frissel. 2000. "Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities." *Conservation Biology* 14 (1): 18–30. doi: 10.1046/j.1523-1739.2000.99084.x.

Varnes, D.J. 1984. *Landslide Hazard Zonation: A Review of Principles and Practice*. Paris: United Nations Educational.

Verkerk, P.J., M. Lindner, G. Zanchi, and S. Zudin. 2011. "Assessing Impacts of Intensified

Biomass Removal on Deadwood in European Forests.” *Ecological Indicators* 11 (1): 27–35.doi: 10.1016/j.ecolind.2009.04.004.

Vikolainen, V., H. Bressers, and K. Lulofs. 2013. “The Role of Natura 2000 and Project Design in Implementing Flood Defence Projects in the Scheldt Estuary.” *Journal of Environmental Planning and Management* 56 (9): 1359–1379. doi: 10.1080/09640568.2012.

Ward Jr., J.H. 1963. “Hierarchical Grouping to Optimize an Objective Function.” *Journal of American Statistical Association* 58 (301): 236–244. doi: 10.1080/01621459.1963.10500845.

Wathern, P. 1988. *Environmental Impact Assessment: Theory and Practice*. New York: Routledge.

Watkins, R.Z., J. Chen, J. Pickens, and K.D. Brosofske. 2003. “Effects of Forest Roads on Understorey Plants in a Managed Hardwood Landscape.” *Conservation Biology* 17 (2): 411–419. doi: 10.1046/j.1523-1739.2003.01285.x.

Wolf, I.D., G. Hagenloh, and D.B. Crof. 2012. “Visitor Monitoring Along Roads and Hiking Trails: How to Determine Usage Levels in Tourist Sites.” *Tourism Management* 33 (1): 16–28. doi: 10.1016/j.tourman.2011.01.019.

Zambelli, P., C. Lora, R. Spinelli, C. Tattoni, A. Vitti, P. Zatelli, and M. Ciolli. 2012. “A GIS Decision Support System for Regional Forest Management to Assess Biomass Availability for Renewable Energy Production.” *Environmental Modelling and Software* 38: 203–213. doi: 10.1016/j.envsoft.2012.05.016.

4. Paper III: Using forest management to control invasive alien species: helping implement the new European regulation on invasive alien species[3]

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Abstract

On 1 January 2015 a new European regulation on invasive alien species entered in force. Key aspects of this regulation are the adoption of a list of invasive alien species which are of European Union concern, the requirement for specific prevention measures, the establishment of early detection and fast eradication measures, and the management of the widely spread invasive alien species. We highlight the potential contribution of the forestry sector to promote the implementation of this regulation. There is a wealth of experience on positive and negative responses of invasive alien species to forestry interventions. This knowledge should be synthesized and further developed to help prevent and manage invasions in forests and adjacent habitats and to minimize the risks of invasive alien species. We thus recommend that decisions regarding the application of the regulation will include actors responsible for, or involved in, the management and use of forests and related semi-natural habitats.

Keywords Forestry, Silviculture, Invasive alien species, European legislation, Impact assessment, Biodiversity conservation, Invasion ecology

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4.1 Introduction

“Will threat of biological invasions unite the European Union?” asked Hulme et al. (2009) in a policy perspective 5 years ago. The answer to this question is the new European regulation on invasive alien species. On 22 October 2014, the European Parliament and Council adopted regulation (EU) No 1143/2014 on the prevention and management of the introduction and spread of invasive alien species (4.11.2014 Official Journal L 317/35, hereinafter regulation), which entered in force on 1 January 2015. This regulation foresees the identification of a list of invasive alien species of particular concern and underlines the importance of prevention, early warning, rapid response followed by eradication and control measures (Genovesi et al. 2015). Significant optimism has been generated by this legislation, which has been awarded the epithets “a long-awaited legislation” (Carboneras et al. 2013), “ambitious” (Beninde et al. 2015), and “innovative” (Genovesi et al. 2015).

The European forestry sector may be critically affected by this new regulation, as forestry has been identified as an important introduction pathway for invasive alien tree species (Rejmánek 2014; Richardson et al. 2014; Richardson and Rejmánek 2011). Nevertheless, not all alien species are invasive, and indeed forestry has benefited from some economically important tree species introductions (Dickie et al. 2014; van Wilgen and Richardson 2014). At the same time, tree invasions may also conflict with silvicultural goals, as these species (e.g. the alien species *Ailanthus altissima* and *Prunus serotina*; Knüsel et al. 2015; Starfinger et al. 2003) can outcompete or impact the growth of more economically-valuable native tree species . This is why the regulation has been also viewed as a “call to silviculturists” (Sitzia 2014).

Forestry is an important sector in Europe, since forests and other wooded lands cover 42 % of European Union land area (Eurostat Press Office 2008), and therefore has a major impact on both environmental and socio-economic conditions. Moreover, in some Member States, such as Italy, the planning and management of forests also covers semi-natural habitats, such as pastures, meadows, and waters (Cullotta et al. 2014) which host habitats and species protected by the Habitats and Birds Directives. The experiences of years of forest planning and management around Europe may be of considerable assistance in addressing the issue of invasive alien species, through the implementation of appropriate forestry activities, to face the issue of invasive alien species. The European forestry sector could thus help achieve the aims of the regulation which deal with both forests and non-forest environments.

A discussion paper (European Commission 2008) which framed the strategy for dealing with invasive alien species in Europe highlighted the multiple roles played by the forestry sector in relation to biological invasions as an introduction pathway, as well as the resulting economic losses and (critically) the potential management measures which could be taken.

In this paper we illustrate opportunities for the European forestry sector to contribute to the implementation of the new regulations on invasive alien species by building on deep-rooted experience in forest management. We argue for an urgently needed shift from viewing and addressing forestry as an important driver of biological invasions (Essl et al.

2010; McConnachie et al. 2015) to involving the forestry sector as a partner in integrated strategic management approaches (van Wilgen and Richardson 2014). Therefore, we stress that forest management and planning should be considered as an adaptive tool to help successfully reduce the problem of invasive alien species (Fig. 1).

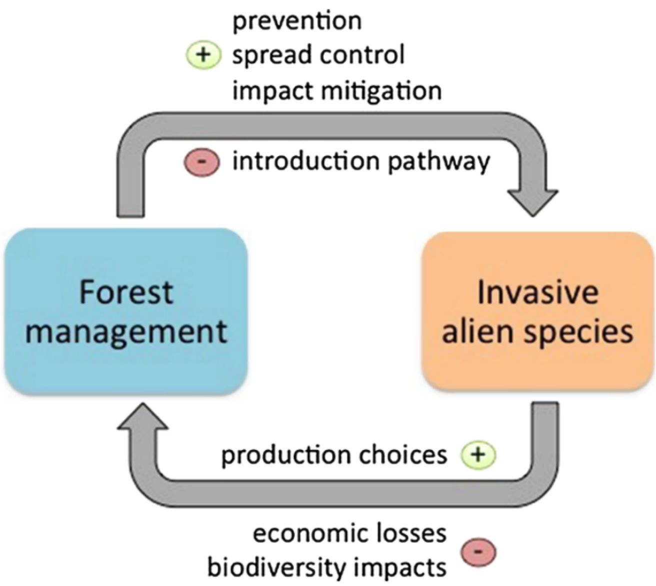

Fig. 1 A conceptual diagram of the links between forest management and invasive alien species. Invasive alien species can widen the array of choices for forest management, as well as hamper it, leading to economic and biodiversity losses. Forest management can help (+) prevent and/or control the spread of invasive alien species, and mitigate their impacts. However, invasive alien species expansion can also be triggered (-) by some types of forest management

4.2 Invasive alien tree species in Europe

According to Rejmánek and Richardson (2013) there are 73 invasive alien tree species in Europe, of which 28 have been used in forestry. An analysis of the most worrying alien species from the main European databases, DAISIE (www.europe-aliens.org) and EPPO (Brunel et al. 2010, www.eppo.int), indicates that 4 tree species (*Acacia dealbata*, *A. altissima*, *P. serotina*, and *Robinia pseudoacacia*) are of particular concern. Among these, the most widely spread is *R. pseudoacacia* (Fig. 2), as it occurs in 42 out of 49 regions in Europe (Lambdon et al. 2008), covers thousands of square kilometres, is of particular forestry interest, and can provide a number of different goods and services (Cierjacks et al. 2013). For example, it has high potential for bioenergy generation with short rotation coppices (Böhm et al. 2011). Based on these characteristics, *R. pseudoacacia* meets the definition of ‘‘widely spread invasive alien species’’ provided by art. 3 of the regulation. However, given the beneficial uses of this species, it is a challenging question as to whether it should be generally banned for silvicultural uses across the European Union, or if it may be used in situations where containment of plantings is feasible. Another prominent tree species of concern in this respect is *A. dealbata*, which invades natural habitats in Portugal, Spain, France, and Italy (Hernández et al. 2014) and has been identified as having high impacts on biodiversity and ecosystem functioning (González-

Muñoz et al. 2012).

Fig. 2 An example of Robinia pseudoacacia invasion in a grassland in the pre-Alpine region of northern Italy

In order to be included in the list of EU concern, species must also meet the criteria listed in art. 4 of the regulation: (a) be alien; (b) be viable and spreading; (c) have a significant adverse impact on biodiversity or the related ecosystem services; (d) that, based on the results of a risk assessment, require concerted action; (e) produce adverse impact that could be effectively prevented, minimised, or mitigated. Indeed, of all alien tree species in Europe, only a limited number would meet these criteria, for example, among the 54 alien conifer species in Europe, less than ten can be considered invasive (Carrillo-Gavilán and Vilà 2010). In this respect, the European Commission funded a study to highlight the minimum standards for a sound risk assessment to include species in the final list of invasive alien species of Union concern (Roy et al. 2014). The authors also screened a tentative list of 80 species, of which five are tree species that may dominate stands (Acer negundo, *Acer rufinerve*, *A. altissima*, *P. serotina*, and *R. pseudoacacia*). Nevertheless, additional tree species do also meet the requirements of art. 4 of the regulation. For example, *Acacia longifolia* has the ability to spread in several European countries (i.e. Portugal, Spain, France, Italy) and can have a significant adverse impact on biodiversity and ecosystem services (Marchante et al. 2008; Rascher et al. 2011).

Moreover, several other woody species, including shrubs or perennial vines species, such as *Akebia quinata*, *Baccharis halimifolia*, *Buddleja davidii*, *Cornus sericea*, *Cotoneaster horizontalis*, *Mahonia aquifolium*, *Pueraria lobata*, and *Rosa rugosa*, are of particular concern (Roy et al. 2014). Although these species mostly occur in habitats outside dense forests, forestry may be involved in their management when the habitats are spatially linked to forests or are otherwise managed by forest authorities. One example is *Amorpha* *fruticosa*, which is found in a range of southeast European and Mediterranean countries (e.g. Hungary, France, and Italy) and can adversely affect coastal, floodplain, and wetland habitats (Quezel et al. 1990; Schnitzler et al. 2007).

4.3 Forest management: risks and opportunities

By taking appropriate measures, the forestry sector has the opportunity to become a major player in curbing future alien tree invasions. Prevention is one of the most important possible interventions foreseen by this new regulation, particularly given the role of the forestry sector as a potential introduction pathway. Indeed, the biological traits that make a tree species good for wood production are also often those that make it a good invader (Dodet and Collet 2012). While the new regulation will enable banning the use and trading of those species entered in the list of Union concern, it would nevertheless be important to carefully carry out scientific screenings of ‘newcomer’ species (Davis et al. 2010), regardless of whether or not they are in the list, and also by applying what has been learnt from past failures and achievements worldwide (Richardson and Blanchard 2011). Indeed, species that are recognised as invasive outside Europe should be treated with particular caution when considering their introduction (Rejmanek 2014). Also of critical importance for the establishment of future prevention measures is the development of a code of conduct for plantation forestry and invasive alien trees, which is currently under preparation on behalf of the Bern Convention (Brundu and Richardson 2015). This document will also highlight reference measures with respect to awareness and containment, as well as to early detection and rapid intervention of invasive alien tree species, in the context of forest plantations.

There is broad evidence that silvicultural practices can either enhance or hamper biological invasions (Table 1). For example, planting alien species such as *P. serotina* or *R. pseudoacacia* for uses such as wind breaks, biomass production, or fire protection and erosion control increases the probability of invasion (Cierjacks et al. 2013; Starfinger et al. 2003). Management practices such as clear-cuts, gap formation, and coppicing can also foster the rejuvenation of some invasive alien tree species (Chabrerie et al. 2008; Hernández et al. 2014; Radtke et al. 2013; Vanhellemont et al. 2010), or favour the spread of invasive herbs such as *Fallopia japonica* (Schnitzler and Muller 1998). However, silviculturists have a long tradition of managing canopy cover, and tree density diameter, and height distribution to change interspecies competition in favour of desired tree species. This experience can be applied to management methods which locally suppress unwanted regeneration of invasive trees or herbs. For example, shelterwood or selection systems of management which are applied in a close-to-nature silvicultural manner are a promising way to reduce invasion risks while preserving the counterpart native communities (Sitzia et al. 2012). These systems exclude clear cuts, artificial regeneration, herbicides, and fertilizer; but even so they can produce significant economic and social added value to ecosystem services (Piussi and Farrell 2000). Even the simple maintenance of continuous tree cover through the implementation of suitable silvicultural systems can prevent the spread of invasive alien plant species. *B. halimifolia*, for example, which is a threat to the understory of alluvial forests and is listed on the tentative list, seems to be outcompeted by native woody species, given an intact vegetative cover (Caño et al. 2013). Restoration of forest communities in floodplains can also help control this species, as well as others.

Table 1 Example of silvicultural measures aimed at reducing the spread of four main invasive alien tree species in Europe

Species	Example of silvicultural measures
Acacia dealbata	Avoid clear cuts and openings [1]
	Maintain or facilitate closed canopy and dense forest [1, 2]
Ailanthus altissima	Avoid coppicing [3]
	Cutting seed plants, underplanting or seeding of shade-tolerant favour native species [4]
Prunus serotina	Avoid clear cuts and openings [4-6]
	Underplanting or seeding of shade-tolerant native species [4, 5]
	Aging with absence of treatments [6, 7]
	Maintain or facilitate closed canopy and dense forest, favour native species [8]
	Girdling [9]
	Single-tree selection systems or group selection systems [4, 8]
Robinia pseudoacacia	Avoid coppicing [3]
	Coppice aging [6, 10]
	Favour native species, conversion from coppice to high forest [4]
	Avoid clear cuts and openings [4-6]
	Girdling [11]

Silvicultural measures are divided into broad types The numbers in superscript refer to the following references: [1] Hernández et al. (2014), [2] Silva and Marchante (2012), [3] Radtke et al. (2013), [4] Regione Piemonte (2013), [5] Skowronek et al. (2013), [6] Terwei et al. (2013), [7] Starfinger et al. (2003), [8] Annighöfer et al. (2015), [9] Annighöfer et al. (2012), [10] Motta et al. (2009), [11] Böcker and Dirk (2008)

The regulation specifically refers to measures such as eradication, population control, containment, and the restriction of their trade. Yet only a limited number of eradication attempts in Europe have been successful (Genovesi 2005), which matches with the limited success worldwide in eradicating invasive alien tree species (van Wilgen and Richardson 2014). In both the EU and worldwide, measures that are frequently applied in the forestry sector are not always successful, and in particular mechanical control measures have been the least successful, and may induce vigorous vegetative regeneration (Annighöfer et al. 2012; Kowarik and Schepker 1998; Skowronek et al. 2013). The array of possible silvicultural measures that could help in contrasting alien tree invasions and the specificity of each case (Simberloff 2014) requires a systematic valuation of their efficiency across different regions and ecosystem types. Furthermore, appropriate silvicultural measures applied to native forest habitats can help in maintaining or improving their resistance to alien species invasions (Jactel et al. 2009). Indeed, the European forestry sector has substantial experience regarding which treatments are suitable to ensure the perpetuation of forests in semi-natural habitats.

Given the wealth of experience at local to national scales on how invasive alien species respond to a range of forestry interventions, we argue that this knowledge should be a central component in implementing the goals of the new European regulation on invasive alien species. This will require:

1. Identifying and avoiding practices that foster the regeneration or spread of invasive alien species.

2. Ascertaining and further testing silvicultural measures that help prevent invasions and control invasive alien species.

3. Sharing knowledge on risks and opportunities of certain measures, with specification for different target species and their environmental contexts throughout Europe.

These approaches are straightforward but run the risk of resulting in only short-term effects, if they are not linked with existing systems of forest planning and management. For this reason, involving forest authorities whenever possible will improve the chance of meeting long-term objectives. This involvement is especially relevant because the relative performance of different management strategies of tree invasion can be influenced by the land-use matrix of the surrounding region (Caplat et al. 2014). For example, management plans which include forests and open habitats can help improve the conservation management of these habitat types, as opposed to considering them individually. The goal of achieving long-term success in managing invasive alien species matches with that of advanced forest planning instruments, which aim at long-term improvements in ecosystem service provision at both the stand and landscape-level, across a range of spatial scales, from a single forest compartment to an entire district. Forest planning is thus challenged to integrate approaches towards managing invasive alien species (Richardson et al. 2014), and this integration would profit from collective learning processes (Secco et al. 2011).

4.4 Conclusion

The implementation of the new European regulation on invasive alien species will require better cooperation among Member States, as well as improved citizens awareness and responsibility (Genovesi et al. 2015). Here we have highlighted the potential contribution of the European forestry sector for promoting the efficient and effective implementation of this regulation, and for controlling the spread of invasive alien species and their associated impacts within Europe in general. There is a wealth of evidence on both the positive and negative effects of forest practices on invasive species that needs to be validated for different environmental contexts. This knowledge must also be made accessible to help prevent and manage invasions within forests, as well as in adjacent ecosystems. We thus recommend that the Working Group on Invasive Alien Species set up by the European Commission’s Directorate, as well as the competent authorities in charge of applying this regulation, and those involved in the public participation will include actors responsible for, or involved in, the management and use of forests and related semi-natural habitats.

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4.5 References

Annighöfer P, Schall P, Kawaletz H, Mölder I, Terwei A, Zerbe S, Ammer C (2012) Vegetative growth response of black cherry (*Prunus serotina*) to different mechanical control methods in a biosphere reserve. Can J For Res 42:2037–2051

Annighöfer P, Kawaletz H, Terwei A, Mölder I, Zerbe S, Ammer C (2015) Managing an invasive tree species—silvicultural recommendations for black cherry (*Prunus serotina* Ehrh). Forstarchiv 86:139–152

Beninde J, Fischer ML, Hochkirch A, Zink A (2015) Ambitious advances of the European Union in the legislation of invasive alien species. Conserv Lett 8:199–205

Böcker R, Dirk M (2008) Development of an effective girdling method to control *Robinia pseudoacacia* L.: first results and outlook. Neobiota 7:63–75

Böhm C, Quinkenstein A, Freese D (2011) Yield prediction of young black locust (*Robinia pseudoacacia* L.) plantations for woody biomass production using allometric relations. Ann For Res 54:215–227

Brundu G, Richardson DM (2015) Code of conduct on plantation forestry and invasive alien trees—First draft—T-PVS/Inf (2015) 1. Council of Europe, Strasbourg

Brunel S, Branquart E, Fried G et al (2010) The EPPO prioritization process for invasive alien plants. EPPO Bull 40:407–422

Caño L, Campos JA, García-Magro D, Herrera M (2013) Replacement of estuarine communities by an exotic shrub: distribution and invasion history of *Baccharis halimifolia* in Europe. Biol Invasions 15:1183–1188

Caplat P, Hui C, Maxwell BD, Peltzer DA (2014) Cross-scale management strategies for optimal control of trees invading from source plantations. Biol Invasions 16:677–690

Carboneras C, Walton P, Vila` M (2013) Capping progress on invasive species? Science 342:930–931

Carrillo-Gavilán MA, Vilà M (2010) Little evidence of invasion by alien conifers in Europe. Divers Distrib 16:203–213

Chabrerie O, Verheyen K, Saguez R, Decocq G (2008) Disentangling relationships between habitat conditions, disturbance history, plant diversity, and American black cherry (*Prunus serotina* Ehrh.) invasion in a European temperate forest. Divers Distrib 14:204–212

Cierjacks A, Kowarik I, Joshi J, Hempel S, Ristow M, von der Lippe M, Weber E (2013) Biological flora of the British Isles: *Robinia pseudoacacia*. J Ecol 101:1623–1640

Cullotta S, Bončina A, Carvalho-Ribeiro SM, Chauvin C, Farcy C, Kurttila M, Maetzke FG (2014) Forest planning across Europe: the spatial scale, tools, and inter-sectoral integration in land-use planning. J Environ Plann Manag 58:1384–1411

Davis AS, Cousens RD, Hill J, Mack RN, Simberloff D, Raghu S (2010) Screening bioenergy feedstock crops to mitigate invasion risk. Front Ecol Environ 8:533–539

Dickie IA, Bennett BM, Burrows LE et al (2014) Conflicting values: ecosystem services and invasive tree management. Biol Invasions 16:705–719

Dodet M, Collet C (2012) When should exotic forest plantation tree species be considered as an invasive threat and how should we treat them? Biol Invasions 14:1765–1778

Essl F, Moser D, Dullinger S, Mang T, Hulme PE (2010) Selection for commercial forestry determines global patterns of alien conifer invasions. Divers Distrib 16:911–921

European Commission (2008) Developing and EU framework for invasive alien species: Discussion paper (Final). European Commission, Brussels

Eurostat Press Office (2008) Forest covers 42% of the EU27 land area. Eurostat News Releases 146/2008:1-3. http://ec.europa.eu/eurostat/web/products-press-releases/-/1-20102008-BP. Accessed 25 July 2015

Genovesi P (2005) Eradications of invasive alien species in Europe: a review. Biol Invasions 7:127–133

Genovesi P, Carboneras C, Vilà M, Walton P (2015) EU adopts innovative legislation on invasive species: a step towards a global response to biological invasions? Biol Invasions 17:1307–1311

González-Muñoz N, Costa-Tenorio M, Espigares T (2012) Invasion of alien *Acacia dealbata* on Spanish *Quercus robur* forests: impact on soils and vegetation. For Ecol Manag 269:214–221

Hernández L, Martínez- Fernández J, Cañellas I, de la Cueva AV (2014) Assessing spatio-temporal rates, patterns and determinants of biological invasions in forest ecosystems. The case of Acacia species in NW Spain. For Ecol Manag 329:206–213

Hulme PE, Pyšek P, Nentwig W, Vilà M (2009) Will threat of biological invasions unite the European Union? Science 324:40–41

Jactel H, Nicoll BC, Branco M et al (2009) The influences of forest stand management on biotic and abiotic risks of damage. Ann For Sci 66:701

Knüsel S, Conedera M, Rigling A, Fonti P, Wunder J (2015) A tree-ring perspective on the invasion of *Ailanthus altissima* in protection forests. For Ecol Manag 354:334–343

Kowarik I, Schepker H (1998) Plant invasions in northern Germany: human perception and response. In: Starfinger U, Edwards K, Kowarik I, Williamson M (eds) Plant invasions: ecological mechanism and human response. Backhuys Publishers, Leiden, pp 109–120

Lambdon P, Pyšek P, Basnou C et al (2008) Alien flora of Europe: species diversity, temporal trends, geographical patterns and research needs. Preslia 80:101–149

Marchante E, Kjøller A, Struwe S, Freitas H (2008) Short-and long-term impacts of *Acacia longifolia* invasion on the belowground processes of a Mediterranean coastal dune ecosystem. Appl Soil Ecol 40:210–217

McConnachie MM, van Wilgen BW, Richardson DM, Ferraro PJ, Forsyth T (2015)

Estimating the effect of plantations on pine invasions in protected areas: a case study from South Africa. J Appl Ecol 52:110–118

Motta R, Nola P, Berretti R (2009) The rise and fall of the black locust (*Robinia pseudoacacia* L.) in the ‘‘Siro Negri’’ Forest Reserve (Lombardy, Italy): lessons learned and future uncertainties. Ann For Sci 66:410

Piussi P, Farrell EP (2000) Interactions between society and forest ecosystems: challenges for the near future. For Ecol Manag 132:21–28

Quezel P, Barbero M, Bonin G, Loisel R (1990) Recent plant invasions in the circum-Mediterranean region. In: di Castri F, Hansen AJ, Debussche M (eds) Biological Invasions in Europe and the Mediterranean Basin, vol 65. Monographiae Biologicae. Springer, Netherlands, pp 51–60

Radtke A, Ambraß S, Zerbe S, Tonon G, Fontana V, Ammer C (2013) Traditional coppice forest management drives the invasion of *Ailanthus altissima* and *Robinia pseudoacacia* into deciduous forests. For Ecol Manag 291:308–317

Rascher KG, Große-Stoltenberg A, Máguas C, Meira-Neto JAA, Werner C (2011) *Acacia longifolia* invasion impacts vegetation structure and regeneration dynamics in open dunes and pine forests. Biol Invasions 13:1099–1113

Regione Piemonte (2013) Le specie vegetali esotiche – Black list - Management list (Gestione). Gruppo di lavoro specie esotiche della Regione Piemonte, Torino, Italy. http://www.regione.piemonte.it/ambiente/tutela_amb/esoticheInvasive.htm Accessed 25 July 2015

Rejmánek M (2014) Invasive trees and shrubs: where do they come from and what we should expect in the future? Biol Invasions 16:483–498

Rejmánek M, Richardson DM (2013) Trees and shrubs as invasive alien species—2013 update of the global database. Divers Distrib 19:1093–1094

Richardson DM, Blanchard R (2011) Learning from our mistakes: minimizing problems with invasive biofuel plants. Curr Opin Environ Sustain 3:36–42

Richardson DM, Rejmánek M (2011) Trees and shrubs as invasive alien species—a global review. Divers Distrib 17:788–809

Richardson DM, Hui C, Nunez MA, Pauchard A (2014) Tree invasions: patterns, processes, challenges and opportunities. Biol Invasions 16:473–481

Roy H, Schonrogge K, Dean H et al (2014) Invasive alien species—framework for the identification of invasive alien species of EU concern (ENV.B.2/ETU/2013/0026). European Commission DG Environment, Brussels

Schnitzler A, Muller S (1998) Ecology and biogeography of highly invasive plants in Europe: giant knotweeds from Japan (*Fallopia japonica* and *F. sachalinensis*). Rev Ecol-Terre Vie 53:3–38

Schnitzler A, Hale BW, Alsum EM (2007) Examining native and exotic species diversity in European riparian forests. Biol Conserv 138:146–156

Secco L, Pettenella D, Gatto P (2011) Forestry governance and collective learning process in Italy: likelihood or utopia? For Policy Econ 13:104–112

Silva J, Marchante H (2012) Post-fire management of exotic forests. In: Moreira F, Arianoutsou M, Corona P, De las Heras J (eds) Post-Fire Management and Restoration of Southern European Forests, vol 24. Managing Forest Ecosystems. Springer, Netherlands, pp 223–255

Simberloff D (2014) Biological invasions: what's worth fighting and what can be won? Ecol Eng 65:112–121

Sitzia T (2014) A call to silviculturists for a new field of science: the forestry of invasive alien species. For Chron 90:486–488

Sitzia T, Campagnaro T, Dainese M, Cierjacks A (2012) Plant species diversity in alien black locust stands: a paired comparison with native stands across a north-Mediterranean range expansion. For Ecol Manag 285:85–91

Skowronek S, Terwei A, Zerbe S et al (2013) Regeneration potential of floodplain forests under the influence of non-native tree species: soil seed bank analysis in Northern Italy. Restor Ecol 22:22–30

Starfinger U, Kowarik I, Rode M, Schepker H (2003) From desirable ornamental plant to pest to accepted addition to the flora? The perception of an alien tree species through the centuries. Biol Invasions 5:323–335

Terwei A, Zerbe S, Zeileis A, Annighöfer P, Kawaletz H, Mölder I, Ammer C (2013) Which are the factors controlling tree seedling establishment in North Italian floodplain forests invaded by non-native tree species? For Ecol Manag 304:192–203

van Wilgen BW, Richardson DM (2014) Challenges and tradeoffs in the management of invasive alien trees. Biol Invasions 16:721–734

Vanhellemont M, Wuters L, Baeten L, Bijlsma R-J, De Frenne P, Hermy M, Verheyen K (2010) *Prunus serotina* unleashed: invader dominance after 70 years of forest development. Biol Invasions 12:1113–1124

5. Paper IV: Multi-scale analysis of alpine landscapes with different intensities of abandonment reveals similar spatial pattern changes: implications for habitat conservation[4]

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Abstract

The abandonment of traditional anthropogenic activities is an important driver shaping landscape patterns. Therefore, multi-scale pattern analysis over time is needed to identify appropriate scales for biodiversity conservation and monitoring of abandoned landscapes. We compared spatial and temporal changes in a pair of alpine watersheds in Italy (Cajada and Tovanella), which are similar in size, geo-climatic conditions, and land-use histories; but have had divergent anthropogenic abandonment processes since the 1950s. We hypothesize that this divergence has led to corresponding dissimilarities in multi-scale patterns of landscape change. To examine this hypothesis, we analyzed land cover maps from three years (1954, 1980/83, 2006) and described the changes using transition matrices. For each year and watershed, landscape heterogeneity and a set of class-level metrics (i.e. percentage of the landscape, area-weighted mean patch size, patch density, area-weighted mean shape index, edge density, and aggregation index) were also measured at different scales using random sampling techniques, and the results were summarized by using scalograms. Woodland expansion occurred mainly at the expenses of grasslands, meadows, and shrublands. These changes were greater during the first time-period (1954-80/83) than in the more recent period (1980/83-2006), with a mean annual value that decreased from +5.18 to +1.33 ha/year and from +4.08 to +1.96 ha/year in the abandoned and managed watersheds, respectively. Landscape heterogeneity decreased over time with a similar pattern in both watersheds, which indicates a general process of homogenization. Management regime affected the spatial-scale response of class-level metrics; these metrics showed a variety of multi-scalar responses, which were not always consistent over time and under different management regimes. When considering the response of the indices across spatial-scales for both watersheds, certain historical curves showed a scale break, representing a significant change in the shape and slope of the curve (i.e. scale divergence). The presence of scale breaks in the scalograms can potentially reveal important thresholds for biodiversity. For example, grassland and meadow patch density at small spatial scales (<200 m radius), which was found to be

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important for protected butterfly species, had a greater reduction over time in the managed watershed when compared to the abandoned watershed. In conclusion, the findings of this study indicate that there is good potential for understanding changes in landscape patterns under different management abandonment regimes by combining spatial and temporal analysis of class-level metrics.

Keywords: Landscape pattern, landscape metrics, reforestation, Natura 2000, multi-scalar, multi-temporal

5.1 Introduction

Anthropogenic activities tend to modify the heterogeneous scales and patterns of natural landscapes (Turner et al., 2013), and therefore a corresponding change in scale and pattern is expected when anthropogenic pressure decreases. Consideration of multiple spatial scales is fundamental to understand spatial complexity (Wu et al., 2011) and to define landscapes through the use of landscape indicators (Lustig et al., 2015). Indeed, the identification of appropriate spatial scales of analysis is a critical step in environ-mental and biodiversity monitoring (Mairota et al., 2015). The effect of changes in grain size (i.e. spatial resolution of data; Turner et al., 1989a) have been widely examined (Frazier, 2016), however there is still the need to further analyze the effect of extent (i.e. size of the study area; Turner et al., 1989a) on landscape indicators (Lustig et al., 2015; Símová and Gdulová, 2012), and to develop an improved understanding of the implications of these changes.

Application of the findings of multi-scale assessments into practical management actions is recognized as an outstanding challenge (Nash et al., 2014). For example, given that landscapes tend to show distinct patterns at different spatial scales, the findings from a single-scale analysis may often be overly reductive (Zurlini and Girardin, 2008). In addition, not all metrics used to quantify landscape pattern respond consistently at different spatial scales. Metrics can be classified depending on whether or not their response is consistent (Wu, 2004), and not all metrics have shown consistent responses among different studies (Símová and Gdulová, 2012). Research on the response of landscape metrics to changes in spatial scale, in particular when coupled with temporal changes under different anthropogenic pressure intensity, can help shed light on impacts to biodiversity (e.g., Frate et al., 2015; Riiters et al., 1997). In this context, a comparison of pattern metrics between different areas will be most valid when the spatial extent is the same and the proportion of land-use categories are approximately equal (Baldwin et al., 2004; Remmel and Fortin, 2013; Turner et al., 2001). Furthermore, coupling the analysis of changes in spatial scale to different time periods may provide further indications on the variable responses of landscape metrics, and what this variability implies about ecological impacts.

Traditional agricultural and forestry practices have altered natural heterogeneous landscapes in many rural mountain areas, resulting in a complex mosaic of sparse open areas and woodland patches. Conversely, many landscapes around the world are now changing again as a consequence of the abandonment of these practices (e.g., Haddaway et al., 2014; Mukul and Herbohn, 2016; Navarro and Pereira, 2012). Management abandonment in mountain landscapes leads to natural succession processes, which typically results in shrub and woodland encroachment (e.g., Chemini and Rizzoli, 2003; Dullinger et al., 2003; MacDonald et al., 2000).

In a review of studies on the impacts of rural abandonment, Sitzia et al. (2010) found general trends towards an increase in size and number of woodland patches and a decrease of open semi-natural habitats linked to anthropogenic activities (e.g., meadows and pastures). The ecological consequences of these changes may be either positive or negative, depending on the geographic and economic context, and the spatial-scale of analysis. Tree encroachment usually results in a simplification and homogenization of these landscapes (Bracchetti et al., 2012), with a decrease in landscape diversity and a reduction in complex mosaics (Frate and Carranza, 2013; Frate et al., 2014; Geri et al., 2010 Frate et al., 2014; Geri et al., 2010). Another potential consequence is the reduction in ecological connectivity across open semi-natural habitats, such as meadows and pasturelands (Sitzia and Trentanovi, 2011).

Analyses of landscape pattern change related to land abandonment typically consider a single scale and assume a dichotomous representation of the landscape by focusing on forest habitats (Otero et al., 2015). However, a broader focus (i.e. studying changes of different land covers) can provide a better understanding of the implications on biodiversity conservation. Analyzing landscape pattern change at various spatial scales also enables a better differentiation of landscapes with different management regimes (Garcia-Feced et al., 2010), and can also help expand our understanding of the complex patterns resulting from land abandonment (Frate et al., 2014). Making comparisons between different landscapes with similar spatial dimensions and geographic conditions, but with different management, would be beneficial in multi-scale analysis (e.g., Bracchetti et al., 2012; Martinez del Castillo et al., 2015; Pan et al., 1999). However much of the previous research on this topic has either focused only on a single study site, has compared sites with very different landscapes (e.g. in terms of area/characteristics), or has not considered landscape metrics (e.g., Beilin et al., 2014; Hall et al., 2012; Tasser et al., 2007).

Changes in landscape composition and configuration due to management abandonment can affect biodiversity both positively and negatively (Queiroz et al., 2014). Navarro and Pereira (2012) highlighted the likely positive effects derived from forest expansion due to farmland abandonment. For example, they suggest that reduced anthropogenic pressure and forest restoration could favor approximately 60 bird, 24 mammal and 26 invertebrate species in Europe. Furthermore, forest expansion increases the area of land suitable for forest species, as some are shade-tolerant plants (Carranza et al., 2012), as well as for some vertebrate species (Bracchetti et al., 2012). Nevertheless, species well suited to semi-natural open habitats may be negatively impacted by the expansion of forests. For example, in the Central Massif region of France, the abundance of open-habitat adapted birds which are of conservation concern decreased in the vicinity of forest edges (Fonderflick et al., 2013). Little attention has been given to landscape features' configuration in relation to their possible suitability as habitat for wildlife within the context of forest expansion (Bowen et al., 2007). It is understood however that shape irregularity metrics correlate with wildlife and vegetation diversity (Carranza et al., 2012; Saura et al., 2008). Yet, analyzing how landscapes change over different spatial extents can better inform scientists about the possible effects of anthropogenic activities on plants and wildlife (Holland et al., 2004; Morelli et al., 2013; Schindler et al., 2013).

The objective of this study is to examine how landscape metrics vary at different scales. Specifically, we aim to analyze spatio-temporal changes in complex landscapes with high biodiversity, which have had different amounts of recent anthropogenic pressure. We examined two forested watersheds with similar sizes and environmental conditions, but with differing management intensity: (1) low-intensity management (gradual abandonment), and (2) no management (abrupt abandonment due to forestry and pasture cessation). We hypothesize that (i) the loss of open habitat types (e.g. grasslands and meadows) due to woodland encroachment will be greater in the abandoned watershed; (ii) management abandonment will smooth the disturbance re-scaling and re-shaping effects occurring in the historic managed watershed, and we should expect a shift from a landscape pattern with pronounced scale-breaks in contrast to a landscape with linear scaling relations (Frate et al., 2014); (iii) after a strong initial divergence between patterns and scales, the differences in landscape metrics between the two areas will tend to disappear; and (iv) landscape metrics for grasslands and meadows, and shrublands will show similar responses, but differing from that observed for woodlands within the same watershed. To our knowledge, this is the first study investigating the responses of landscape indicators to changes in spatial scale coupled with temporal and management regime changes. These changes in landscape pattern at different scales can potentially have effects for habitat and species; therefore we discuss the results of this study in the context of their implications for biodiversity conservation.

5.2 Materials and Methods

5.2.1. Study area

This study is based in the Tovanella and Cajada watersheds (1040 ha each), located within the south-eastern Alps in the Veneto Region of Italy, in the Alpine biogeographical region (Fig. 1). The two watersheds are located less than 6 km from each other. The climate is temperate-continental, typical of the south-eastern alpine region, with relatively high mean annual precipitation (1300–1500 mm year^{-1}) concentrated in May-June and October-November, and a mean annual temperature of 7.2°C with harsh winters. The main rock substrate is dolomitic limestone formed during the secondary and tertiary. Both watersheds have an altitudinal range from approximately 550 to 2500 m a.s.l.

While the historical management regime of both watersheds is characterized by traditional forestry and pastoral activities, their management trajectories diverged after the 1950s. The forests in both watersheds were heavily logged by the Republic of Venice between the 15th and 17th centuries, which relied on timber from this region for ships construction. More recently, timber extraction carried out between 1943 and 1953 resulted in a very low growing stock (<200 m3 ha^{-1}) in these forests (Susmel, 1958). In addition to forestry, the pastures and meadows of both watersheds were important for the pastoral activities of local communities. For example, in Tovanella in the second half of the 14th century, around one hundred cattle and thousands of sheep and goats grazed the pastures and meadows during the summer season (Viola et al., 2008).

After the 1950s, the management approach of the two watersheds diverged significantly. In Tovanella, forestry and pasture activities were abruptly abandoned, and when the area became an ‘Oriented Biogenetic Nature Reserve’ in 1971, all anthropogenic activities were legally banned (Viola et al., 2008). By contrast, in Cajada forestry has continued at a low intensity (i.e. near-to-nature silviculture, applying group shelterwood system) and pastoral activities have continued at gradually decreasing rate, upuntil the present time period (Cassol, 1996).

Both watersheds are in the Natura 2000 network, Tovanella falls under the 'Site of Community Importance (SCI): Val Tovanella –Bosconero – IT3230031' and the 'Special Protection Area (SPA):Dolomiti del Cadore e Comelico – IT3230089', while Cajada falls under the SPA and SCI 'Dolomiti Feltrine e Bellunesi – IT3230083'.The establishment of this protection underlines the importance of both areas as habitats for biodiversity conservation (Table 1)and species of Community interest (Ente Parco Nazionale Dolomiti Bellunesi, 2009; Lasen et al., 2008). Both watersheds are dominated by woodlands, consisting mainly of beech and fir, which were classified as *Asperulo-Fagetum* beech forests (code: 9130), following the Habitats Directive (Directive 92/43/EEC) classification. Calcareous rocky slopes with chasmophytic vegetation (code: 8210) are widespread in the higher altitudes, while brush-land areas of *Pinus mugo* and *Rhododendron hirsutum* (code: 4070) are common in Tovanella. Grassland habitats are less prevalent, and the most common categories are alpine and subalpine calcareous grasslands (code: 6170). Several flora and fauna species of European interest are present, including those related to forests (e.g., *Glaucidium passerinum* L., *Cypripedium calceolus* L.), open grasslands (e.g., *Parnassius mnemosyne* L., *P. apollo* L.), ecotones and mosaics (e.g., *Tetrao tetrix* L., *Lanius collurio* L.), and rocky slopes and screes (e.g., *Campanula morettiana* Rchb., *Physoplexis comosa* (L.) Schur.) (Argenti and Lasen, 2008; Ente Parco Nazionale Dolomiti Bellunesi, 2009; Hardersen and Dal Cortivo, 2008; Mezzavilla et al., 2008).

5.2.2 Land cover maps

To characterize the changes in the two watersheds, we used aerial photographs for the years 1954 (flight from the Italian Military Geographical Institute-GAI), 1980 (flight from Aerofoto Consult), 1983 (flight from Rossi srl), and 2006 (flight from Regione Veneto). While the photos of the latter year were already orthorectified and georeferenced (TIFF and ECWimages), those of the former years were acquired in paper format, and then georeferenced and digitalized in a Geographic Information System. The aerial photos from 1980 (for Tovanella) and 1983 (for Cajada) were scanned as TIFF images with a resolution of 800 dpi, and the 1954 photos were scanned at 1200 dpi. The resolution for these photos were selected based on the clarity of the output, as settings at higher resolutions resulted in grainy images All photos were orthorectified using the ErMapper 7.0 software with a 25 m Digital Terrain Model (DTM). For the images from 1980 and 1983, a minimum of 10 ground control points were used for orthorectification, with a resulting average root-mean-square error (RMSE) of 1.15 and 0.81 m for 1980 and 1983, respectively. As the calibration certificate was missing for 1954 images, the spline method was applied by using an average of 20 points for each photo from previously georefenced images. To produce land-cover maps, a manual classification process was carried out. A classification grid with a mapping unit of 250 m^2 (15.8 × 15.8 m) at a fixed scale of 1:5000 was used. This resolution enabled consideration of a 1 mm minimal possible mapping accuracy, which corresponds to 5 m at a scale 1:5000 (Sitzia and Trentanovi, 2011). Six cover classes were used: forests, grasslands and meadows, shrublands, bare rock, buildings, and alpine grasslands (i.e., grasslands above the forest line). We selected these cover classes as they represent the most important habitats for the plant and wildlife species of interest for this study.

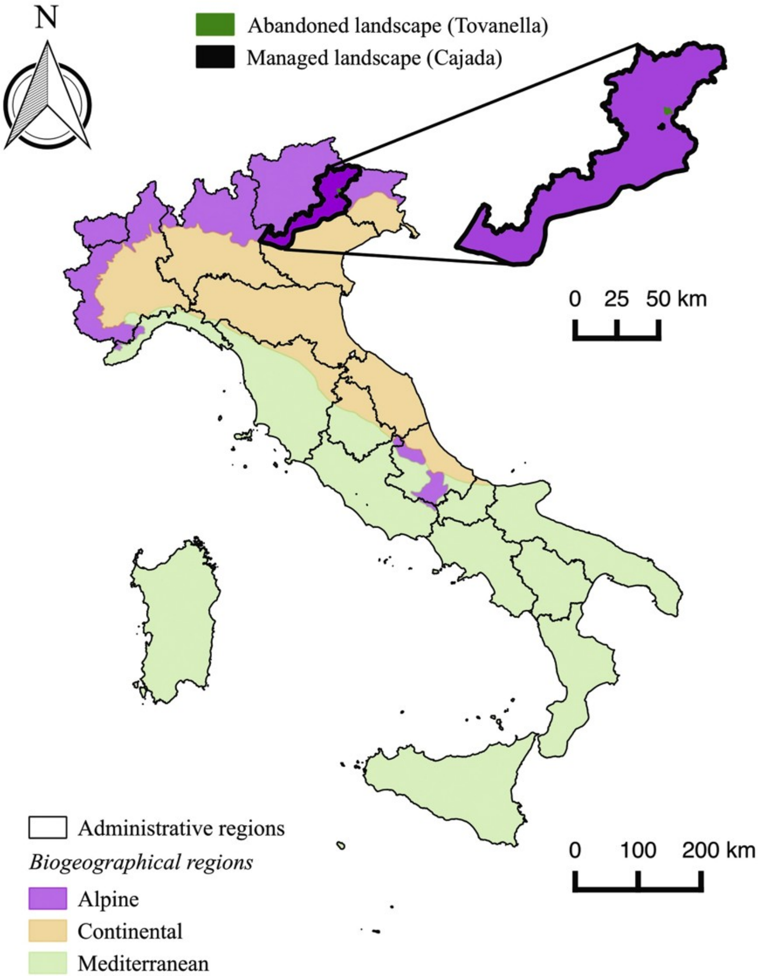

Figure 1: Study area location within the alpine biogeographical region in Veneto (north-eastern Italy).

5.2.3. Data analysis

5.2.3.1. Landscape scale dynamics

To analyse the main land-cover changes between the periods under investigation (1954–1980/83 and 1980/83-2006), we built specific transition matrices using the ‘combine tool’ of the GIS software ‘ArcGIS 10.1’ (ESRI, 2011). The transition matrices describe the temporal dynamics of the analysed watersheds over each time period (1954–1980/83 and 1980/83-2006). Each cell of the matrix represents the hectares belonging to one land cover class in a given year that has changed into another land cover class. The diagonal cells represent the unchanged (or persistence) area. For each period we calculated the mean annual change (ha/year).

5.2.3.2. Spatial pattern analysis at multiple scales

To quantify the spatial pattern changes across scales, a set of landscape indices were selected and computed using the software 'FRAGSTATS 4.2' (McGarigal et al., 2012). These metrics were calculated at the landscape level (taking into account all cover types together) and at the class level considering three cover types (woodlands, grasslands and meadows, and shrublands) that host habitats (Table 1) and species of Community interest. Landscape and class pattern indices were selected due to their ability to relate the observed landscape pattern to the underlying ecological processes. Furthermore, these metrics were previously reported as ecologically meaningful and have proven useful for describing and comparing the spatial structure of abandoned land (Algeet-Abarquero et al., 2015; Frate et al., 2014; Otero et al., 2015; Schindler et al., 2013).

The Shannon diversity index (SHDI) was used to define landscape heterogeneity (Díaz-Varela et al., 2009b) and to detect possible homogenization processes as a consequence of land management abandonment. Class level indices, as a percentage of the landscape (PLAND), area-weighted mean patch size (AREA_AM), and patch density (PD) were analyzed to assess changes in the extent, patch size, and spatial distribution of each cover type. As the active management of forests and pastures should allow the maintenance of open habitats, management abandonment should promote a progressive reduction in the number and size of these habitats (e.g. Rocchini et al., 2006). Area-weighted mean shape index (SHAPE_AM), edge density (ED), and aggregation index (AI) were computed to observe changes in shape and connectivity of each cover type. Indeed, forest expansion usually leads to the oversimplification of patch structure (i.e. more regular) and to a decrease of open habitat connectivity (Sitzia et al., 2010). The description of the pattern metrics used in the study along with their respective variation range is provided in McGarigal et al. (2012).

Since landscape patterns and processes are scale-dependent (e.g., Turner et al., 1989b; Wu, 2004; Wu et al., 2002), and landscape indices vary with landscape extent (Frate et al., 2014; Gardner et al., 1987; Wu et al., 2002), we quantified landscape pattern over time at multiple scales. There are several techniques available for landscape analysis at multiple scales, including nested quadrat design (e.g. Turner et al., 1989b), diagonal expansion of the study area (e.g., Frate et al., 2014; Wu, 2004; Wu et al., 2002), step-wise expansion of the original area (e.g. Baldwin et al., 2004), grid-based sampling design (Sitzia et al., 2014) and moving window analysis (e.g. Díaz-Varela et al., 2009a). Here we adopted a sampling strategy (Carranza et al., 2014; Ramezani et al., 2013; Stehman, 2012), which provides a good method of describing the relationship between land cover and spatial pattern changes (Carranza et al., 2014; Díaz-Varela et al., 2009b), which is crucial for the correct interpretation of on-going landscape processes (Frate et al., 2014; Hargis et al., 1998). Moreover, a sample-based approach allows for the production of statistically valid estimates of class and landscape metrics at different scales (Hassett et al., 2012). In particular we quantified landscape pattern change using random sampling techniques on the multi-temporal maps. One hundred points were randomly distributed across the land cover map, and circular windows with different radii at increasing dimensions were used for each point. Selected radii were 100, 200, 300, 500, 700 and 1000 m. In this way, each window defined a series of sub-landscapes on which the selected pattern metrics were computed. Scalograms for all indices were built by plotting index value against spatial scales, and a simple spline regression model (with the relative bootstrapped 95% confidence interval) was fitted to evaluate the response of the indices (the shape and the slope of the regression curve) and their changes over time (e.g. small-scale vs. large-scale changes).

Table 1: Habitats of Annex I Habitats Directive (Directive 92/43/EEC) and related land-cover classes in the two watersheds (Tovanella and Cajada).

Cover class	HD Habitats	Watershed
Alpine grasslands	6210 Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (* important orchid sites)	Cajada
Alpine grasslands, or Grasslands and meadows	6170 Alpine and subalpine calcareous grassland	Cajada and Tovanella
Grasslands and meadows	6430 Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels	Cajada and Tovanella
	6520 Mountain hay meadows	Cajada
	7230 Alkaline fens	Cajada
Shrublands or Woodlands	4060 Alpine and Boreal heaths	Tovanella
Shrublands	4070 * Bushes with *Pinus mugo* and *Rhododendron hirsutum* (*Mugo-Rhododendretum hirsuti*)	Cajada and Tovanella
Rocks and screes	8120 Calcareous and calcshist screes of the montane to alpine levels (*Thlaspietea rotundifolii*)	Cajada and Tovanella
	8210 Calcareous rocky slopes with chasmophytic vegetation	Cajada and Tovanella
Woodlands	9130 *Asperulo-Fagetum* beech forests	Cajada and Tovanella
	9140 Medio-European subalpine beech woods with *Acer* and *Rumex arifolius*	Cajada and Tovanella
	9150 Medio-European limestone beech forests of the *Cephalanthero-Fagion*	Tovanella
	9180 * *Tilio-Acerion* forests of slopes, screes and ravines	Tovanella
	91K0 Illyrian *Fagus sylvatica* forests (*Aremonio-Fagion*)	Cajada
	9420 Alpine *Larix decidua* and/or *Pinus cembra* forests	Tovanella
	9530 * (Sub-)Mediterranean pine forests with endemic black pines	Cajada and Tovanella

5.3 Results

5.3.1 Landscape change

The temporal maps of the two watersheds (Fig. 2) indicate that significant changes have occurred in the whole area over the last 50 years. In 1954, woodlands were the dominant land cover for both watersheds, followed by shrublands, and grasslands and meadows. During the first time span in Cajada, a steady decrease in grasslands and meadows (from 8% to 4%), and shrublands (from 11% to 7%) was observed, along with an increase in woodlands (from 63% to 76%). The corresponding mean annual change was −1.37, −1.36 and +4.08 ha/year, respectively. During the second time-span, grasslands and meadows, and shrublands showed a slight decrease (from 4% to 3% and from 7% to 6%) whereas woodlands increased from 76% to 81%. However, the mean annual change was lower (−0.27, −0.49, and +1.96 ha/year, respectively) than during the previous period. In Tovanella watershed, from 1954 to 1980 the area covered by grasslands and meadows decreased from 4% to 1%, shrublands decreased from 28% to 21%, and woodlands increased from 49% to 62%. This corresponds to a mean annual change of −1.09 ha/year for grasslands and meadows, −2.89 ha/year for shrublands, and +5.18 ha/year for woodlands. In the period 1980–2003, grasslands and meadows almost disappeared, shrublands decreased from 21% to 19%, and woodlands expanded from 62% to 65%. The mean annual change was lower compared to the period 1954–1980, corresponding to −0.16 ha/year for grasslands and meadows, −0.24 ha/year for shrublands, and +1.33 ha/year for woodlands.

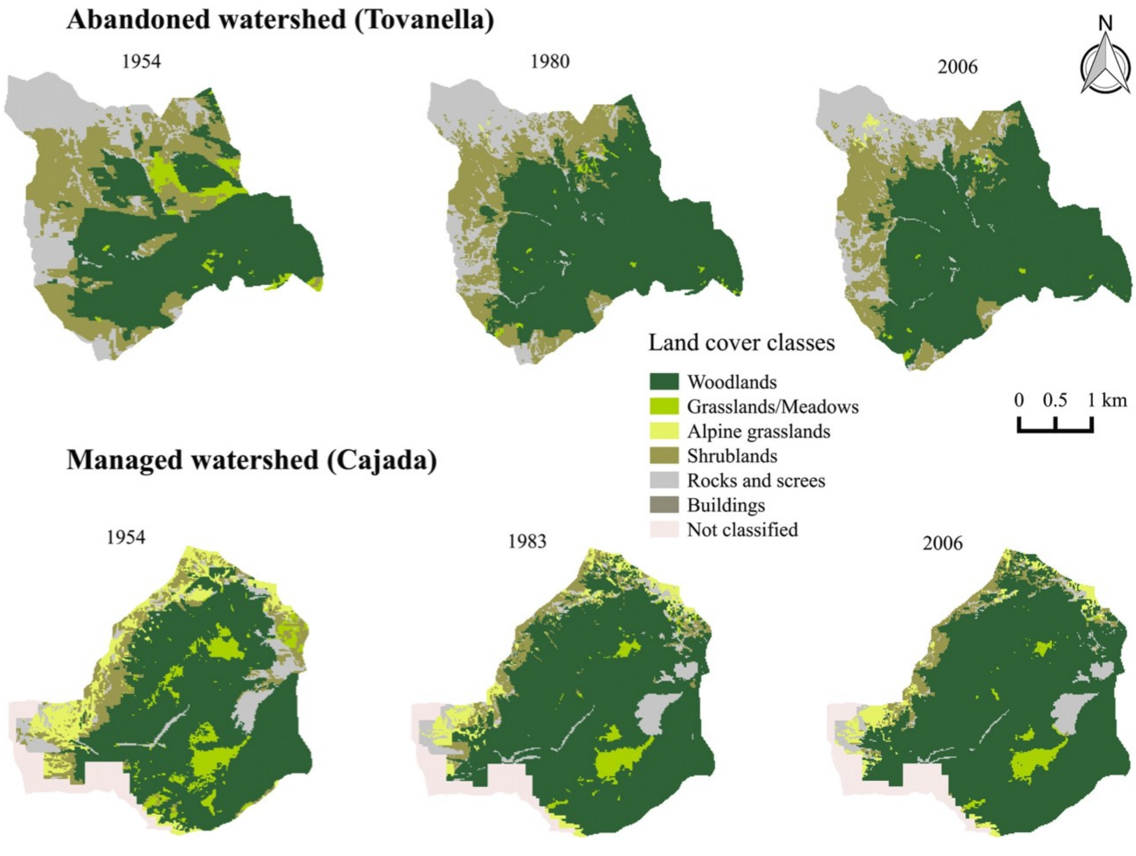

Figure 2: Land-cover maps of the abandoned (above) and managed (below) watersheds for the different analyzed years (1954, 1980/83, 2006).

The transition matrices (Table 2 and Table 3) and the relative maps of change (Fig. 3) show that during the first time period both watersheds had the same percentage of stable (25%) and dynamic areas (75%). In the last time-period, these values differed slightly between watersheds, with an unchanged area of 11% for Cajada and 10% for Tovanella. In the period 1954–1980/1983 woodlands and bare rock showed high values of persistence for both watersheds. Despite these similarities, differences between the two watersheds were observed for other land-cover categories. In Cajada, shrubland was the category most affected by change (mainly conversion into woodlands), whereas in Tovanella, grasslands and meadows showed the greatest change (95% reduction from the original cover). In the second time-period, smaller changes were observed in both watersheds. In Cajada, the largest change was recorded for shrublands, and for grasslands and meadows; while in Tovanella grasslands and meadows, and alpine grasslands were less persistent.

5.3.2 Multi-scale analysis of landscape change

The temporal analysis revealed different patterns emerging at specific scales in the two watersheds. The scalograms of Shannon Diversity Index (SDI) revealed differences in heterogeneity across scales and over time (Fig. 4). Overall, Tovanella had higher heterogeneity values than Cajada. The SDI was almost always higher in 1954, and significantly decreased in the recent years of analysis. However, it is interesting to note that in Tovanella at very local scales there were no differences in the SDI. When looking at the response of the metrics across scales for both areas, the historical curves showed a scale break representing a significant change of the shape and slope of the curve ("scale divergence" Wu et al., 2000). After this point the curves were relatively consistent, and further increasing the scale did not cause substantial variations in the metric value. Conversely, the 2006 curves did not show such a break, but they grew slightly in a linear fashion.

The scalograms describing the class pattern metrics (Figs. 5–7) showed specific responses to changing scale that varied among cover classes, year, and between the study areas. According to the land-cover classes, two main trends can be distinguished for percentage of the landscape (PLAND) across scales: (i) a scale break for grasslands and meadows, and shrublands and (ii) a steady-linear response curve for woodlands. The percentage of grasslands and meadows in 1954 was very similar between Tovanella and Cajada at the local scale, however at larger scales PLAND was higher in Cajada. In 1954, PLAND was characterized by a curve with a sharp scale break (close to 400 m radius) after which the curve was relatively constant, while in the more recent years of analysis PLAND tended to be stable across scales. This means that in 1954 grasslands and meadows formed a significant element of the small-scale patterns, while more recently they were close to disappearing at the local scale as well. Shrublands had a similar response compared to grasslands and meadows, except for in Tovanella where they were still an important factor in landscape heterogeneity. At local scales, shrublands were extensive in all compared years. In 1954, the percentage of shrublands decreased until reaching a break point (at 400 m radius) where the values tended to become constant.

Table 2: Transition matrix of the abandoned watershed (Tovanella) for the two time-periods (1954-1980 and 1980-2006). Areas in bold did not change land cover class.

(a) Time-period 1954-1980								
		1980						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1954
1954	Woodlands	**487.35**	2.40	11.50	4.90	0.00	0.00	506.15 (49)
	Grasslands and meadows	31.18	**2.03**	3.40	0.90	0.00	0.00	37.50 (4)
	Shrublands	108.65	3.95	**153.15**	25.00	0.00	0.10	290.85 (28)
	Rocks and screes	13.58	0.90	47.63	**143.45**	0.00	0.68	206.23 (20)
	Buildings	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00 (0)
	Alpine grasslands	0.00	0.00	0.00	0.00	0.00	**0.00**	0.00 (0)
	Total 1980	640.75 (62)	9.28 (1)	215.68 (21)	174.25 (17)	0.00 (0)	0.78 (0)	1040.73 (100)
(b) Time-period 1980-2006								
		2006						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1980
1980	Woodlands	**632.30**	0.43	6.35	1.68	0.00	0.00	640.75 (62)
	Grasslands and meadows	5.13	**2.33**	1.68	0.15	0.00	0.00	9.28 (1)
	Shrublands	32.43	0.73	**166.70**	15.63	0.00	0.20	215.68 (21)
	Rocks and screes	5.38	1.60	27.35	**135.30**	0.00	4.63	174.25 (17)
	Buildings	0.00	0.00	0.00	0.00	**0.00**	0.05	0.05 (0)
	Alpine grasslands	0.00	0.00	0.13	0.60	0.00	**0.05**	0.78 (0)
	Total 2006	675.23 (65)	5.08 (0)	202.20 (19)	153.35 (15)	0.00 (0)	4.93 (0)	1040.78 (100)

Table 3: Transition matrix of the managed watershed (Cajada) for the two time-periods (1954-1980 and 1983-2006).

(a) Time-period 1954-1983								
		1983						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1954
1954	Woodlands	**556.73**	5.88	5.40	13.75	0.00	4.65	586.40 (63)
	Grasslands and meadows	43.45	**27.25**	2.75	0.75	0.05	0.03	74.28 (8)
	Shrublands	62.33	1.10	**27.20**	6.68	0.00	7.68	104.98 (11)
	Rocks and screes	14.98	0.20	5.78	**47.88**	0.00	2.25	71.08 (8)
	Buildings	0.00	0.00	0.00	0.00	**0.00**	0.00	0.00 (0)
	Alpine grasslands	27.30	0.00	24.50	5.88	0.00	**36.88**	94.55 (10)
	Total 1983	704.78 (76)	34.43 (4)	65.63 (7)	74.93 (8)	0.05 (0)	51.48 (6)	931.28 (100)
(b) Time-period 1983-2006								
		2006						
		Woodlands	Grasslands and meadows	Shrublands	Rocks and screes	Buildings	Alpine grasslands	Total 1983
1983	Woodlands	**693.08**	2.78	5.18	1.20	0.00	2.55	704.78 (76)
	Grasslands and meadows	9.13	**24.93**	0.35	0.03	0.00	0.00	34.43 (4)
	Shrublands	25.78	0.40	**34.63**	1.10	0.00	3.73	65.63 (7)
	Rocks and screes	9.18	0.20	5.30	**53.43**	0.00	6.83	74.93 (8)
	Buildings	0.00	0.00	0.00	0.00	**0.05**	0.00	0.05 (0)
	Alpine grasslands	12.68	0.00	8.90	2.98	0.00	**26.93**	51.4810)
	Total 2006	749.83 (81)	28.30 (3)	54.35 (6)	58.73 (6)	0.05 (0)	40.03 (4)	931.28 (100)

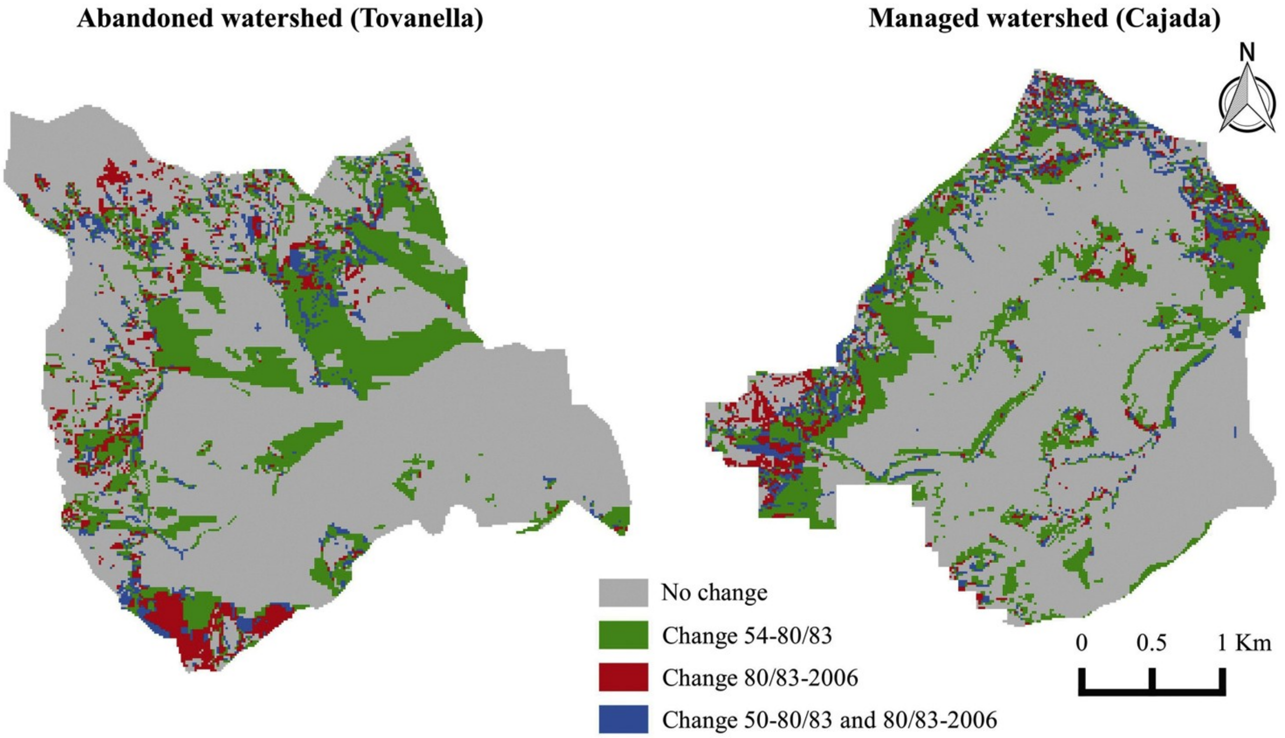

Figure 3: Maps indicating the areas where land cover changes occurred during the first (1954-1980/83 in green), second (1980/83- 2006 in red) and both (1954-1980/83 and 1980/83 -2006 in blue) time-periods. Abandoned watershed is outlined in the left and managed one in the right. Areas where no changes occurred are also reported (grey).

In 1980 and 2006, this percentage decreased faster across scales and the break point occurred at larger scales. This indicates that shrublands have become less prominent, while conversely woodlands did not show any sort of scale break, but were dominant at all scales.

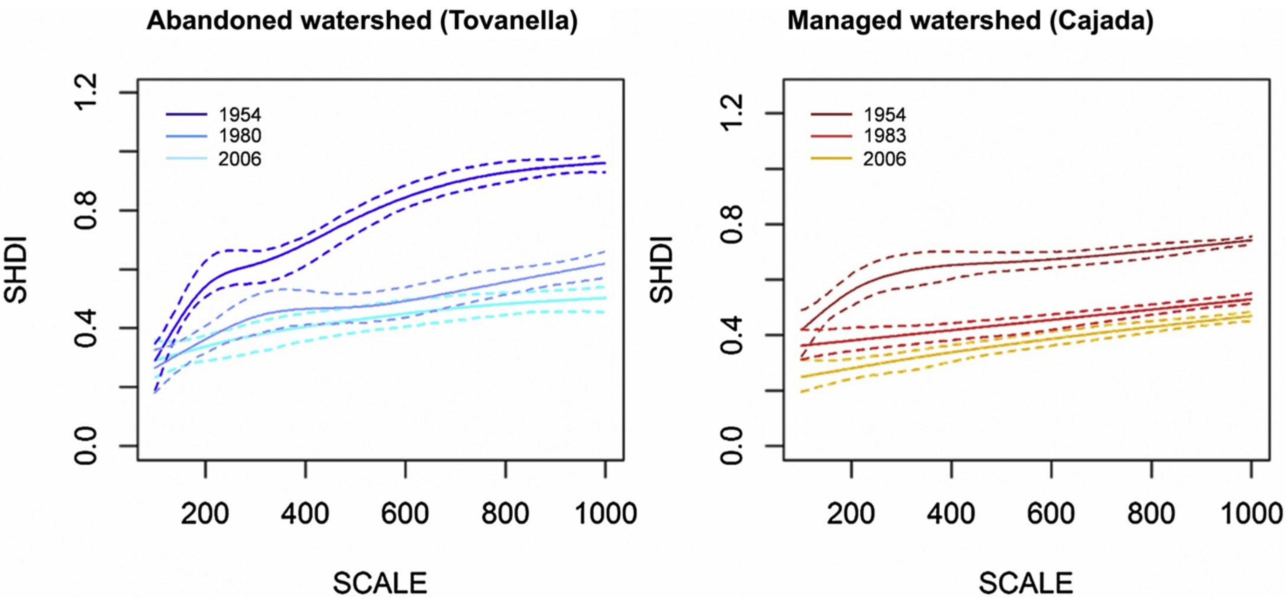

Figure 4: Shannon Diversity Index (SDI) at different scales for the three years (1954, 1980/83, 2006) in abandoned (left) and managed (right) watersheds.

Concerning patch density (PD), all classes showed a similar scalar response characterized by a strong linear-decay pattern at very small scales that tended to stabilize at small scales (200–400 m radius). Thus, the small-scale patchiness was replaced by large-scale cohesion between patches belonging to a specific class. Over the three years there were no differences in patch density across scales, except for Cajada where both grasslands and meadows, and shrublands, had lower patch density values in more recent years.

The area-weighted mean patch area (AREA_MN) showed different scaling relations for grasslands and meadows, and shrublands in the two watersheds. For Tovanella in the 1954 scalogram, grasslands and meadows exhibited a linear-increase trend, whereas the 1980 and 2006 scalograms showed flat curves with values close to zero. This pattern was very similar to that of Cajada shrublands. Conversely, both Tovanella shrublands and Cajada grasslands and meadows had a linear-increase response curve with the 1954 Cajada grasslands and meadows scalogram presenting a scale break over the 700–800 m radius scale. Overall, 1954 curve had higher AREA_MN values. Woodlands exhibited a power-law increasing trend for both areas, with higher values in more recent years.

The aggregation index (AI) showed different scale response according to the land cover classes under consideration. Grasslands and meadows AI had an erratic response, as the metric seemed not to follow any predictable trend. Conversely, shrublands exhibited a flat scalogram with AI values that were always lower in the two recent dates compared to that of the 1954.Woodlands had a similar scale response but without significant difference over time.

Edge density (ED) showed a similar response for grasslands and meadows, and shrublands presenting scale-breaks at different extents. However, scale-breaks for grasslands and meadows tended to disappear in 1980/83 and 2006 compared to 1954. Furthermore, differences in ED were more evident in Cajada between the first year and the other two years of analysis. Whereas, differences between years for shrublands were more evident in Tovanella than in Cajada. Unexpectedly, the scale-break in Tovanella tended to become more evident in recent years while the opposite occurred in Cajada. A change in woodland edge density was evident at all scales, and higher in 1954 in Cajada, while in Tovanella occurred only between 200 and 600 m radii.

Area-weighted mean shape index (SHAPE AM) scalogram for grasslands and meadows, shrublands and woodlands had similar response in Cajada: an increasing trend with spatial extent and decreasing over time except at small scales. Also in Tovanella SHAPE AM increased with increasing extent. For grasslands and meadows it was higher in 1954, except at small scales, while for shrublands it was lower.

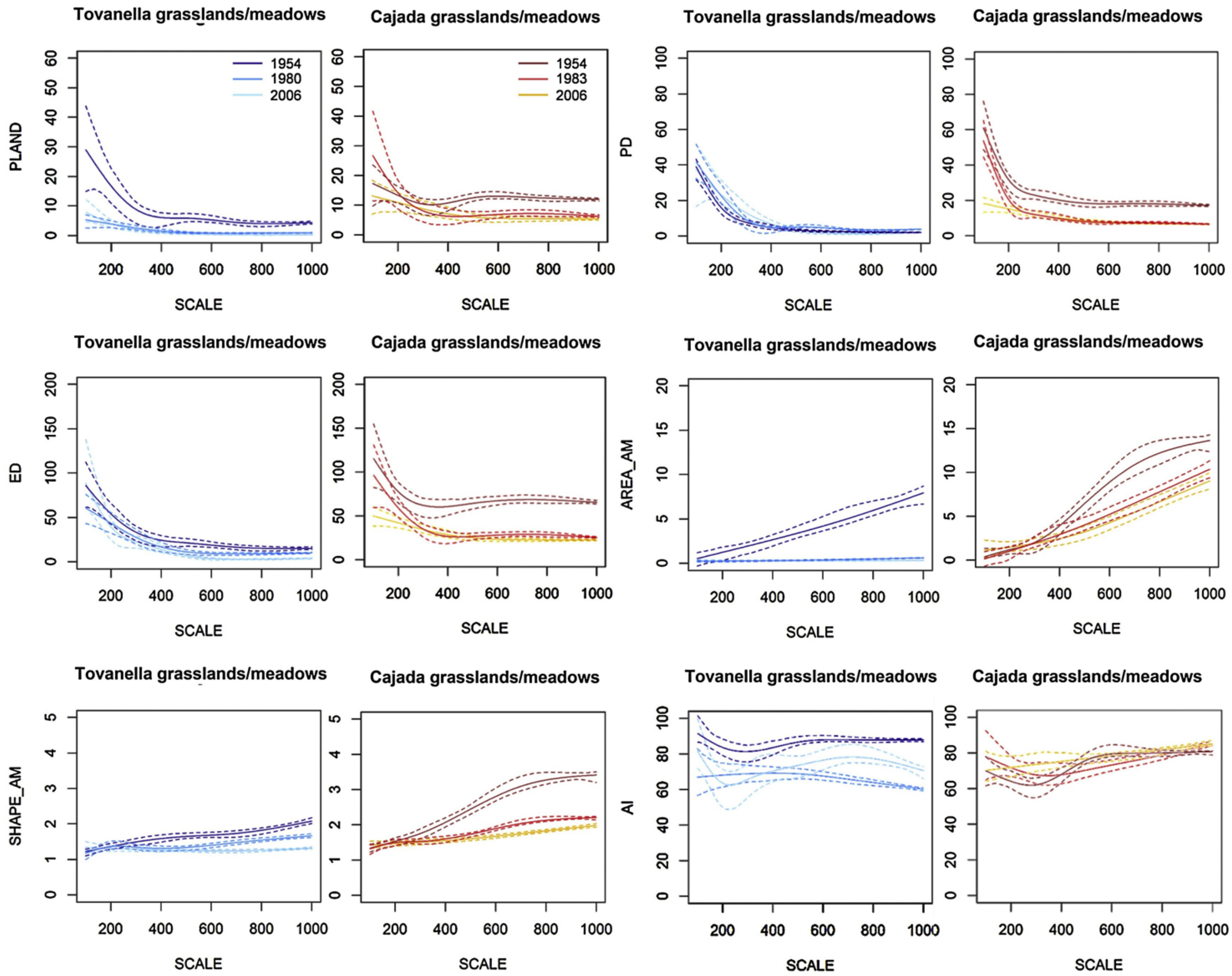

Figure 5: Multi-scale response of landscape metrics (PLAND, PD, ED, AREA_AM, SHAPE_AM, AI) of grasslands and meadows for the three years in abandoned (left) and managed (right) watersheds.

5.4 Discussion

5.4.1 Landscape-scale dynamics

In contrast to our hypothesis, similar results were observed in terms of habitat loss regardless of the differences in the management regime between the two watersheds. Woodland area initially expanded rapidly in the period of 1950-80/83, while in the period 1980/83-2006 the mean annual expansion decreased substantially in both watersheds irrespective of the initial cover. Both watersheds also showed similar trends for grasslands and meadows, and shrublands; which showed a strong initial loss followed by a reduction in the mean annual change over the second period. The pattern is common in the Alps, where the reduction of these land covers has been widely reported as an effect of management abandonment and consequent woodland encroachment (e.g., Orlandi et al., 2016; Sitzia et al., 2010). Climate change has also likely impacted vegetation changes at high altitudes, resulting in an increase in forest cover (Dainese and Sitzia, 2013; Evangelista et al., 2016; Jackson et al., 2016). This trend in cover reduction is likely to have negatively affected species that prefer grasslands and meadows, and shrubland habitats. Furthermore, the various grassland and heath habitats protected under the Habitats Directive found in the two watersheds (i.e. Alpine and Boreal heaths – code: 4060, Alpine and subalpine calcareous grassland – code 6170, Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (* important orchid sites) – code 6210, Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels – code 6430, Mountain hay meadows – code 6520, Alkaline fens – code 7230), are all at least in part dependent on anthropogenic activities (Halada et al., 2011).

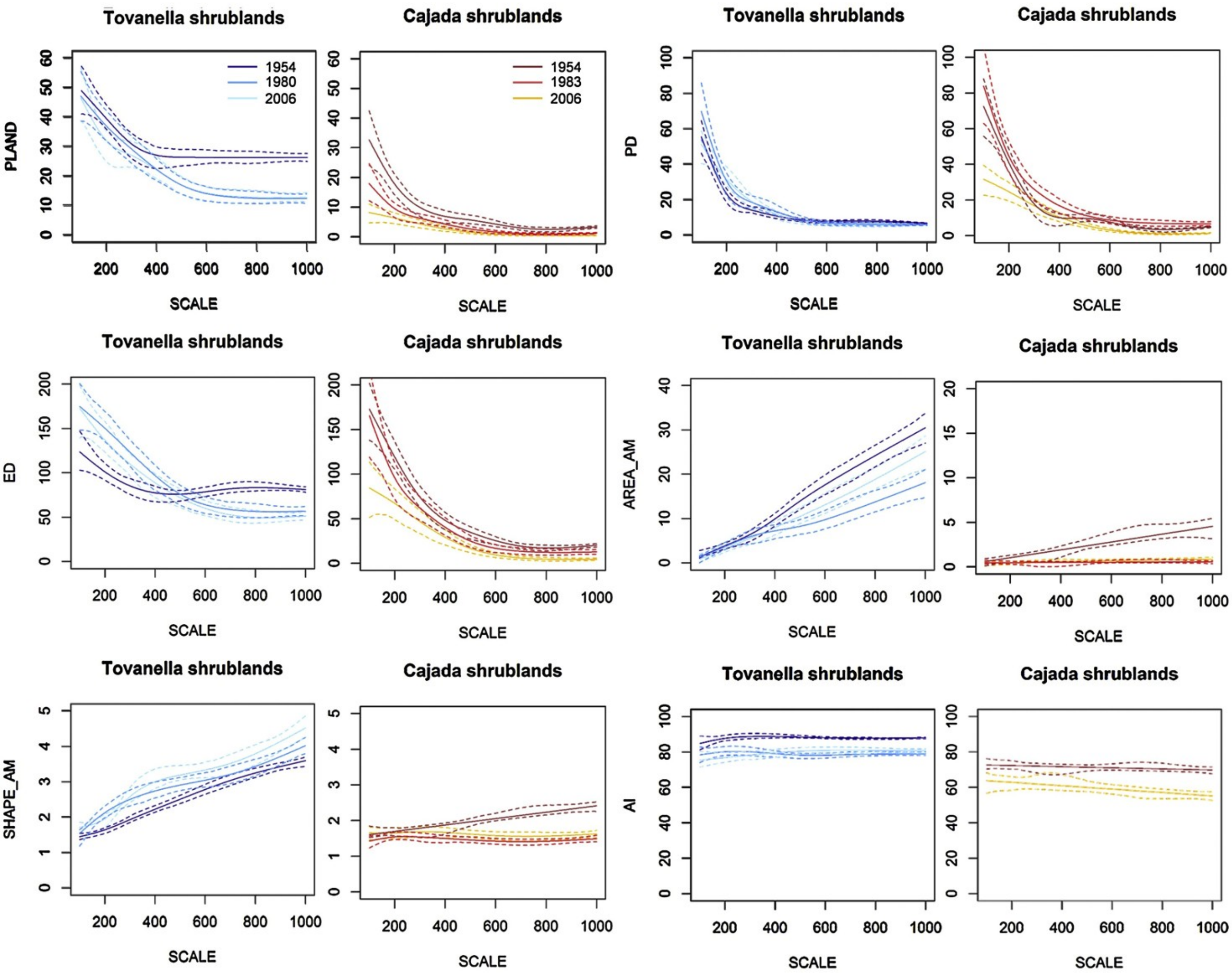

Figure 6: Multi-scale response of landscape metrics (PLAND, PD, ED, AREA_AM, SHAPE_AM, AI) of shrublands for the three years in abandoned (left) and managed (right) watersheds.

It is probable in both watersheds that species linked to these habitats will have moved towards other areas and/or their population will have reduced over time (Pernollet et al., 2015). Indeed, such a trend (i.e. reduction of suitable habitat due to woodland succession) can lead to local extinction of these species (Balmer and Erhardt, 2000; Schlossberg and King, 2009). By contrast, woodland species have probably benefited from these changes (Sirami et al., 2007), as the differences in specific habitat features between the two watersheds have shown an influence on several species (Nascimbene et al., 2013; Sitzia et al., 2015). The analysis of land cover change and related habitat loss over time, as conducted in this study, enables the spatial identification of areas that underwent changes in recent years, which should be preferred areas for restoration actions (Öckinger et al., 2006). However, attention should be given to time lags in specialist species local extinction and woodland specialist colonization (Bagaria et al., 2015)

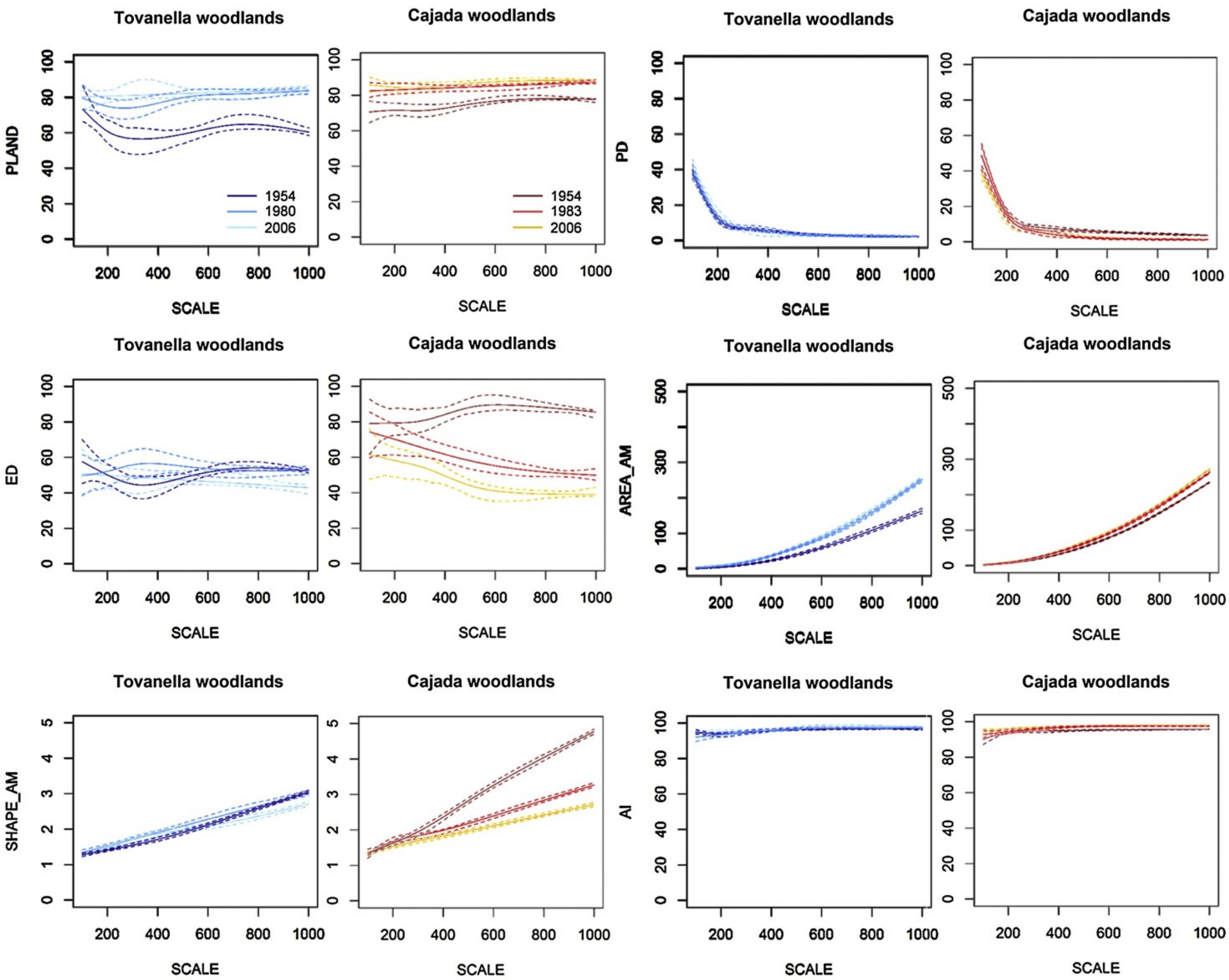

Figure 7: Multi-scale response of landscape metrics (PLAND, PD, ED, AREA_AM, SHAPE_AM, AI) of forest for the three years in abandoned (left) and managed (right) landscapes

5.4.2 Multi-scale response of landscape change

Landscape heterogeneity decreased in both watersheds over time, but with slightly different changes across scales. A decrease in landscape heterogeneity was recorded in other parts of the Alps (Kulakowski et al., 2011). In Tovanella (hereafter called abandoned) watershed the decrease in heterogeneity occurred between 1954 and 1980, and tended to stabilize between 1980 and 2006. In Cajada (hereafter called managed) watershed the decrease was evident between all the three years. Furthermore, in the abandoned watershed there was a marked gradual incorporation of different patches with increasing scale that was less pronounced in the managed watershed for 1954; namely, the curve tended to flatten at lower scales in the managed watershed. This indicates a higher initial heterogeneity in the abandoned areas than in the managed watershed. However, in both areas the observed pattern indicated a similar presence of mainly continuous woodland cover over time (1980/83, 2006) and scales. Indeed, this demonstrates an important reduction of landscape heterogeneity in both the abandoned and managed watersheds. This was particularly evident in the abandoned watershed at larger extents, and in general the reduction of landscape heterogeneity was greater at larger extents in both watersheds.

Wildlife species have various scales of habitat selection (Ducci et al., 2015; Mayor et al., 2009; Sitzia et al., 2014). Species that are “multi-habitat” specialist (i.e. preferring high heterogeneity at broader scales; Russo, 2007), are likely to have been impacted from the reduction in heterogeneity over time in both watersheds given the reduction in landscape heterogeneity over time. For example, bird species are frequently affected by landscape changes at intermediate extents (Mairota et al., 2015), for which we observed a greater reduction in landscape diversity. Heterogeneity was reduced at finer scales in the managed watershed, suggesting that taxa needing high heterogeneity at finer scales were more vulnerable in this location. For example, in a study in Germany, Steckel et al. (2014) found that wasp richness was positively influenced by fine, rather than medium, scale landscape heterogeneity.

Our results suggest that low-intensity management, both with respect to forestry and agro-pastoral activities, was not sufficient for the maintenance of a heterogeneous landscape. It is likely that the low-intensity management applied in the managed watershed was masking a substantial abandonment of agro-pastoral activities, and that forest management alone did not fully prevent tree encroachment. This is particularly relevant for protected areas (e.g. Natura 2000 sites) in which grassland and shrubland habitats are of conservation interest. For example, both long continuity grassland management and current activities are fundamental for high species density (Cousins et al., 2007; Eriksson et al., 2002) which is a common indicator of the conservation status of semi-natural habitats. Furthermore, compositional and configurational heterogeneity support taxonomically diverse butterfly communities and vulnerable species linked to grassland habitats, respectively (Perović et al., 2015). Therefore, in addition to near-to-nature forestry, extensive grazing activities (Cocca et al., 2012) and mowing should be promoted to maintain a degree of landscape heterogeneity, as heterogeneity is likely to have a positive effect on the conservation of habitats and species.

The analysis of relationships between single land covers and the response of landscape metrics over extent and time gives additional insights into the changes occurring in the two watersheds. Understanding the response of class-level metrics at multiple scales is fundamental to characterize and monitor landscape heterogeneity (Wu, 2004). Our results highlight the huge variability in the response of class-level metrics to changes in scale (Kelly et al., 2011; Wu, 2004) and time under different anthropogenic pressure regimes. For example, woodlands had stable, linear-decay, and power-law increasing responses to scale with or without scale-breaks. All metrics examined were sensitive to changes in extent, however certain metrics showed less predictable trends (e.g. aggregation index) than others (e.g. area-weighted mean patch size). Previous studies have described different types of responses to change in extent (e.g., Baldwinet al., 2004; Wu, 2004); however certain metrics can be considered to respond following specific scaling relationships (Símová and Gdulová, 2012).

As for landscape diversity, response of class metrics among scales and time tended to be similar between the abandoned and managed watersheds. Therefore, both watersheds shared similar scaling and shaping dynamics not showing substantial influence over time. Furthermore, while most of the metrics seemed to generally have a consistent response regardless of the considered class (grasslands and meadows, shrubland, woodland), the percentage of landscape and edge density had a different response when we considered woodland, or grasslands and meadows, or shrubland classes. These differences can be linked to the different changes in cover of the three classes and to their different total cover within the whole watershed. Class-level edge density is believed to have an unpredictable response with increasing extent (type III metrics, Wu, 2004), and this seemed to be confirmed in our study. Patch density is also considered to be unpredictable, and in our study it tended to have a decreasing power function, characterized by a strong linear-decay pattern at very small scales in nearly all cases. However, patch density tended to also level at low scales for recent grasslands and meadows, and shrublands in the low-intensity managed watershed. Furthermore, other metrics responded differently that what has been reported in previous studies; for example, Wu (2004) found a staircase-like response with changing extent for area weighted mean shape index, in contrast to the findings of our study. Argañaraz and Entraigas (2014) found that changes in grain size and extent varied for different landscapes and among class types. Furthermore, our results suggest that metrics response depends on the landscape class under investigation, and that the response is also likely to change with changing anthropogenic pressure. Hence, the response of metrics to changes in extent will depend on the underlying processes occurring in the landscape. Observing the response of metrics among different time periods helps in comparing the intensity of change over extent after changes in management.

This interaction between landscape management and the response of class-level metrics of landscape patterns at changing scales holds an important informative function for the conservation of habitats and species. The capacity of landscape metrics to predict species occurrence is affected by the spatial scale of analysis; for example, Schindler et al. (2013) observed that woody plants, orthopterans, and small terrestrial birds are better predicted at smaller extents than reptiles. In our study, differences in the response of metrics over time were more evident at smaller spatial extents. Therefore, species that respond to changes at small scales are likely to have been impacted by the management regime in the two watersheds. Indeed, species are likely to respond to changes in landscape pattern in relation to their home range and dispersal capability. Species with small home ranges and short dispersal distance will be more influenced by changes at small scales, while species with larger home ranges and longer dispersal distance will be more affected by differences at larger scales. For example, *P. mnemosyne*, a butterfly of conservation concern found in the study region, is dependent on grassland habitats. The dispersal distance of this species is considered to be relatively small (253 m ±12.59), indicating that patch density at small distances is important for their migration (Välimäki and Itämies, 2003). In our case study, grasslands and meadows patch density at small scales (<200 m radius) slightly decreased over time in the abandoned watershed, whereas in the low-intensity man-aged watershed it underwent a drastic decrease over time. This phenomenon, together with a decrease at smaller scales of the percentage of the landscape and a more disperse pattern of this habitat type, indicates a strongest reduction of suitable conditions for this species in the low-intensity managed watershed during the last time period examined. Similarly, other vulnerable species may have experienced changes occurring at small/medium spatial scales. For example, *L. achine* has a relatively low dispersal distance (<500 m) and females favour the edges of woodland openings for laying eggs (Bergman, 1999; Bergman and Landin, 2002); therefore it is likely that changes in edge density at small scales (i.e. stronger in our managed watershed) had an influence on the communities of this species. Furthermore, species such as *Tetrastes bonasia* L. may have responded to different features at small spatial scales (Sitziaet al., 2014). Nevertheless, species with longer dispersal distance may have not responded to these small-scale changes, but rather at those occurred at larger scales; for example *Alectoris graeca saxatilis* Meisner, which has an average dispersal distance of 4–15 km (Bernard-Laurent, 1991; Cattadori et al., 2003).

5.5 Conclusion

This study compared landscape pattern changes occurring over time and space in one watershed where management was abandoned and in one were management continued over time, but with low-intensity. In both watersheds, woodland cover increased with similar trends (at the expenses of grasslands and meadows, and shrublands). A loss in landscape heterogeneity occurred regardless of the management regimes in place in the two watersheds, primarily between 1954 and 1980/83. The landscape metrics showed a variety of responses depending on scale, time, habitat type, and anthropogenic pressure. Indeed, these complex interactions, as shown by landscape metrics, highlight the importance of taking into account multiple perspectives for characterizing different landscapes (Lustig et al., 2015).

Our study indicates that management regime can affect the spatial scale response of landscape and class-level metrics. A reduction in scale breaks for grasslands and meadows, and shrublands over time highlighted the relevant spatial changes. Understanding the changes in response of specific landscape metrics over scale, time, habitat type, and management regime are important, as they have implications for biodiversity conservation, especially for species that may be sensitive to habitat modification (Sitzia and Trentanovi, 2011). Monitoring landscape metrics have the potential to help the assessment of the conservation status of habitats under the Habitats Directive (Perrino et al., 2013; Vaz et al., 2015) and this should be further investigated in other landscape settings taking into account relevant habitat characteristics. Our study highlighted that the landscape response of grasslands and meadows, and shrublands was similarly affected by abandonment but also by low-intensity management. These results suggest that the local extinction of many species linked to grasslands and meadows, and shrubland habitats, may have occurred in both watersheds; prob ably earlier in the abandoned than in the low-intensity managed watershed. Indeed, future studies should investigate time lags following changes in landscape metric response, in order to adopt conservation measures for habitats of high biodiversity value in a timely and adequate manner (Bagaria et al., 2015).

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5.6 References

Algeet-Abarquero, N., Sánchez-Azofeifa, A., Bonatti, J., Marchamalo, M., 2015. Land cover dynamics in Osa Region, Costa Rica: secondary forest is here to stay. Reg. Environ. Chang. 15, 1461-1472.

Argañaraz, J.P., Entraigas, I., 2014. Scaling functions evaluation for estimation of landscape metrics at higher resolutions. Ecol. Inform. 22, 1-12.

Argenti, C., Lasen, C., 2008. Vascular plants of Val Tovanella Nature Reserve, in: Hardersen, S., Mason, F., Viola, F., Campedel, D., Lasen, C., Cassol, M. (Eds.), Research on the Natural Heritage of the Reserves Vincheto di Celarda and Val Tovanella (Belluno Province Italy), Conservation of two Protected Areas in the Context of a Life Project. Arti Grafiche Fiorini, Verona, pp. 349-361.

Bagaria, G., Helm, A., Rodà, F., Pino, J., 2015. Assessing coexisting plant extinction debt and colonization credit in a grassland–forest change gradient. Oecologia 179, 823-834.

Baldwin, D.J.B., Weaver, K., Schnekenburger, F., Perera, A.H., 2004. Sensitivity of landscape pattern indices to input data characteristics on real landscapes: implications for their use in natural disturbance emulation. Landsc. Ecol. 19, 255-271.

Balmer, O., Erhardt, A., 2000. Consequences of succession on extensively grazed grasslands for Central European butterfly communities: rethinking conservation practices. Conserv. Biol. 14, 746-757.

Beilin, R., Lindborg, R., Stenseke, M., Pereira, H.M., Llausàs, A., Slätmo, E., Cerqueira, Y., Navarro, L., Rodrigues, P., Reichelt, N., Munro, N., Queiroz, C., 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.

Bergman, K.-O., 1999. Habitat utilization by *Lopinga achine* (Nymphalidae: Satyrinae) larvae and ovipositing females: implications for conservation. Biol. Conserv. 88, 69-74.

Bergman, K.-O., Landin, J., 2002. Population structure and movements of a threatened butterfly (*Lopinga achine*) in a fragmented landscape in Sweden. Biol. Conserv. 108, 361-369.

Bernard-Laurent, A., 1991. Migrant rock partridge (*Alectoris graeca saxatilis*) in the southern French Alps. J. Ornithol. 132, 220-223.

Bowen, M.E., McAlpine, C.A., House, A.P.N., Smith, G.C., 2007. Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna. Biol. Conserv. 140, 273-296.

Bracchetti, L., Carotenuto, L., Catorci, A., 2012. Land-cover changes in a remote area of central Apennines (Italy) and management directions. Landsc. Urban Plan. 104, 157-170.

Carranza, M.L., Frate, L., Acosta, A.T.R., Hoyos, L., Ricotta, C., Cabido, M., 2014. Measuring forest fragmentation using multi-temporal remotely sensed data: Three decades of change in the dry Chaco. Eur. J. Remote Sens. 47, 793-804.

Carranza, M.L., Frate, L., Paura, B., 2012. Structure, ecology and plant richness patterns in fragmented beech forests. Plant Ecol. Divers. 5, 541-551.

Cassol, M., 1996. Studio di fattibilità per la tutela e l'utilizzazione dell'area di Palughét. Ente Parco Nazionale Dolomiti Bellunesi, Feltre (BL), Italy, p. 63.

Cattadori, I.M., Ranci-Ortigosa, G., Gatto, M., Hudson, P.J., 2003. Is the rock partridge *Alectoris graeca saxatilis* threatened in the Dolomitic Alps? Anim. Conserv. 6, 71-81.

Chemini, C., Rizzoli, A., 2003. Land use change and biodiversity conservation in the Alps. J. Mt. Ecol. 7, 1-7.

Cocca, G., Sturaro, E., Gallo, L., Ramanzin, M., 2012. Is the abandonment of traditional livestock farming systems the main driver of mountain landscape change in Alpine areas? Land Use Policy 29, 878-886.

Cousins, S.A.O., Ohlson, H., Eriksson, O., 2007. Effects of historical and present fragmentation on plant species diversity in semi-natural grasslands in Swedish rural landscapes. Landsc. Ecol. 22, 723-730.

Dainese, M., Sitzia, T., 2013. Assessing the influence of environmental gradients on seed mass variation in mountain grasslands using a spatial phylogenetic filtering approach. Perspect. Plant Ecol. Evol. Syst. 15, 12-19.

Díaz-Varela, E., Álvarez-López, C.J., Marey-Pérez, M.F., 2009a. Multiscale delineation of landscape planning units based on spatial variation of land-use patterns in Galicia, NW Spain. Landsc. Ecol. Eng. 5, 1-10.

Díaz-Varela, E.R., Marey-Pérez, M.F., Rigueiro-Rodriguez, A., Álvarez-Álvarez, P., 2009b. Landscape metrics for characterization of forest landscapes in a sustainable framework: Potential application and prevention of misuse. Ann. For. Sci. 66, 301.

Ducci, L., Agnelli, P., Di Febbraro, M., Frate, L., Russo, D., Loy, A., Carranza, M.L., Santini, G., Roscioni, F., 2015. Different bat guilds perceive their habitat in different ways: a multiscale landscape approach for variable selection in species distribution modelling. Landsc. Ecol. 30, 2147-2159.

Dullinger, S., Dirnbock, T., Grabherr, G., 2003. Patterns of shrub invasion into high mountain grasslands of the Northern Calcareous Alps, Austria. Arct. Antarct. Alp. Res. 35, 434-441.

Ente Parco Nazionale Dolomiti Bellunesi, 2009. Piano di Gestione del sito Natura 2000 SIC/ZPS IT3230083 Dolomiti feltrine e bellunesi. Ente Parco Nazionale Dolomiti Bellunesi, Feltre (BL), Italy.

Eriksson, O., Cousins, S.A.O., Bruun, H.H., 2002. Land use history and fragmentation of traditionally managed grasslands in Scandinavia. J. Veg. Sci. 13, 743-748.

ESRI, 2011. ArcGIS Desktop: Release 10. Environmental Systems Research Institute, Redlands, CA.

Evangelista, A., Frate, L., Carranza, M.L., Attorre, F., Pelino, G., Stanisci, A., 2016. Changes in composition, ecology and structure of high-mountain vegetation: a re-visitation study over 42 years. AoB Plants 8.

Fonderflick, J., Besnard, A., Martin, J.-L., 2013. Species traits and the response of open-habitat species to forest edge in landscape mosaics. Oikos 122, 42-51.

Frate, L., Acosta, A.T.R., Cabido, M., Hoyos, L., Carranza, M.L., 2015. Temporal change in forest contexts at multiple extents: three dacades of fragmentation in the Gran Chaco (1979-2010), Central Argetina. PloS ONE 10, e0142855.

Frate, L., Carranza, M.L., 2013. Quantifying landscape-scale patterns of temperate forests over time by means of Neutral Simulation Models. Int. J. Geo-Inf. 2, 94-109.

Frate, L., Saura, S., Minotti, M., Di Martino, P., Giancola, C., Carranza, M.L., 2014. Quantifying forest spatial pattern trends at multiple extents: an approach to detect significant changes at different scales. Remote Sens. 6, 9298-9315.

Frazier, A.E., 2016. Surface metrics: scaling relationships and downscaling behavior. Landsc. Ecol. 31, 351-363.

Garcia-Feced, C., Saura, S., Elena-Rossello, R., 2010. Assessing the effect of scale on the ability of landscape structure metrics to discriminate landscape types in Mediterranean forest districts. For. Syst. 19, 129-140.

Gardner, R.H., Milne, B.T., Turner, M.G., O'Neill, R.V., 1987. Neutral models for the analysis of broad-scale landscape pattern. Landsc. Ecol. 1, 19-28.

Geri, F., Rocchini, D., Chiarucci, A., 2010. Landscape metrics and topographical determinants of large-scale forest dynamics in a Mediterranean landscape. Landsc. Urban Plan. 95, 46-53.

Haddaway, N.R., Styles, D., Pullin, A.S., 2014. Evidence on the environmental impacts of farm land abandonment in high altitude/mountain regions: a systematic map. Environmental Evidence 3, 17.

Halada, L., Evans, D., Romão, C., Petersen, J.-E., 2011. Which habitats of Europena importance depend on agricultural practices? Biodivers. Conserv. 20, 2365-2378.

Hall, J.M., Van Holt, T., Daniels, A.E., Balthazar, V., Lambin, E.F., 2012. Trade-offs between tree cover, carbon storage and floristic biodiversity in reforesting landscapes. Landsc. Ecol. 27, 1135-1147.

Hardersen, S., Dal Cortivo, M., 2008. Butterfly communities (Lepidoptera: Rhopalocera) of the clearings of Val Tovanella Nature Reserve: effects of enlarging abandoned moutain meadows, in: Hardersen, S., Mason, F., Viola, F., Campedel, D., Lasen, C., Cassol, M. (Eds.), Research on the Natural Heritage of the Reserves Vincheto di Celarda and Val Tovanella (Belluno Province Italy), Conservation of two Protected Areas in the Context of a Life Project. Arti Grafiche Fiorini, Verona, pp. 415-424.

Hargis, C.D., Bissonette, J.A., David, J.L., 1998. The behavior of landscape metrics commonly used in the study of habitat fragmentation. Landsc. Ecol. 13, 167-186.

Hassett, E.M., Stehman, S.V., Wickham, J.D., 2012. Estimating landscape pattern metrics from a sample of land cover. Landsc. Ecol. 27, 133-149.

Holland, J.D., Bert, D.G., Fahrig, L., 2004. Determining the spatial scale of species' response to habitat. Bioscience 54, 227-233.

Jackson, M.M., Topp, E., Gergel, S.E., Martin, K., Pirotti, F., Sitzia, T., 2016. Expansion of subalpine woody vegetation over 40 years on Vancouver Island, British Columbia, Canada. Can. J. For. Res. 46, 437-443.

Kelly, M., Tuxen, K.A., Stralberg, D., 2011. Mapping changes to vegetation pattern in a restoring wetland: finding pattern metrics that are consistent across spatial scale and time. Ecol. Indic. 11, 263-273.

Kulakowski, D., Bebi, P., Rixen, C., 2011. The interacting effects of land use change, climate change and suppression of natural disturbances on landscape forest structure in the Swiss Alps. Oikos 120, 216-225.

Lasen, C., Scariot, A., Sitzia, T., 2008. Natura 2000 Habitats map, forest types and vegetation outline of Val Tovanella Nature Reserve, in: Hardersen, S., Mason, F., Viola, F., Campedel, D., Lasen, C., Cassol, M. (Eds.), Research on the natural heritage of the reserves Vincheto di Celarda and Val Tovanella (Belluno Province Italy), Conservation of two protected areas in the context of a Life project. Arti Grafiche Fiorini, Verona, pp. 325-334.

Lustig, A., Stouffer, D.B., Roigé, M., Worner, S.P., 2015. Towards more predictable and consistent landscape metrics across spatial scales. Ecol. Indic. 57, 11-21.

MacDonald, D., Crabtree, J.R., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., Gibon, A., 2000. Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. J. Environ. Manag. 59, 47-69.

Mairota, P., Cafarelli, B., Labadessa, R., Lovergine, F., Tarantino, C., Lucas, R.M., Nagedra, H., Didham, R.K., 2015. Very high resolution Earth observation features for monitoring plant and animal community structure across multiple spatial scales in protected areas. Int. J. Appl. Earth Obs. Geoinf. 37, 100-105.

Martinez del Castillo, E., García-Martin, A., Longares Aladrén, L.A., de Luis, M., 2015. Evaluation of forest cover change using remote sensing techniques and landscape metrics in Moncayo Natural Park (Spain). Appl. Geogr. 62, 247-255.

Mayor, S.J., Schneider, D.C., Schaefer, J.A., Mahoney, S.P., 2009. Habitat selection at multiple scales. Ecoscience 16, 238-247.

McGarigal, K., Cushman, S.A., Ene, E., 2012. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available at the following web site: http://www.umass.edu/landeco/research/fragstats/fragstats.html.

Mezzavilla, F., Cassol, M., Lombardo, S., 2008. Birds of Val Tovanella Nature Reserve, in: Hardersen, S., Mason, F., Viola, F., Campedel, D., Lasen, C., Cassol, M. (Eds.), Research on the natural heritage of the reserves Vincheto di Celarda and Val Tovanella (Belluno Province Italy), Conservation of two protected areas in the context of a Life project. Arti Grafiche Fiorini, Verona, pp. 441-447.

Morelli, F., Pruscini, F., Santolini, R., Perna, P., Benedetti, Y., Sisti, D., 2013. Landscape heterogeneity metrics as indicators of bird diversity: determining the optimal spatial scales in different landscapes. Ecol. Indic. 34, 372-379.

Mukul, S.A., Herbohn, J., 2016. The impacts of shifting cultivation on secondary forests dynamics in tropics: A synthesis of the key findings and spatio temporal distribution of research. Environmental Science & Policy 55, Part 1, 167-177.

Nascimbene, J., Dainese, M., Sitzia, T., 2013. Contrasting responses of epiphytic and dead wood-dwelling lichen diversity to forest management abandonment in silver fir mature woodlands. For. Ecol. Manage. 289, 325-332.

Nash, K.L., Allen, C.R., Angeler, D.G., Barichievy, C., Eason, T., Garmestani, A.S., Graham, N.A.J., Granholm, D., Knutson, M., Nelson, R.J., Nyström, M., Stow, C.A., Sundstrom, S.M., 2014. Discontinuities, cross-scale patterns, and the organization of ecosystems. Ecology 95, 654-667.

Navarro, L.M., Pereira, H.M., 2012. Rewilding Abandoned Landscapes in Europe. Ecosystems 15, 900-912.

Öckinger, E., Eriksson, A.K., Smith, H.G., 2006. Effects of grassland abandonment, restoration and management on butterflies and vascular plants. Biol. Conserv. 133, 291-300.

Orlandi, S., Probo, M., Sitzia, T., Trentanovi, G., Garbarino, M., Lombardi, G., Lonati, M., 2016. Environmental and land use determinants of grassland patch diversity in the western and eastern Alps under agro-pastoral abandonment. Biodivers. Conserv. 25, 275-293.

Otero, I., Marull, J., Tello, E., Diana, G.L., Pons, M., Coll, F., Boada, M., 2015. Land abandonment, landscape, and biodiversity: questioning the restorative character of the forest transition in the Mediterranean. Ecol. Soc. 20, 7.

Pan, D., Domon, G., de Blois, S., Bouchard, A., 1999. Temporal (1958–1993) and spatial patterns of land use changes in Haut-Saint-Laurent (Quebec, Canada) and their relation to landscape physical attributes. Landsc. Ecol. 14, 35-52.

Pernollet, C.A., Korner-Nievergelt, F., Jenni, L., 2015. Regional changes in the elevational distribution of the Alpine Rock Ptarmigan Lagopus muta helvetica in Switzerland. Ibis 157, 823-836.

Perović, D., Gámez-Virués, S., Börschig, C., Klein, A.-M., Krauss, J., Steckel, J., Rothenwöhrer, C., Erasmi, S., Tscharntke, T., Westphal, C., 2015. Configurational landscape heterogeneity shapes functional community composition of grassland butterflies. J. Appl. Ecol. 52, 505-513.

Perrino, E.V., Tomaselli, V., Costa, R., Pavone, P., 2013. Conservation status of habitats (Directive 92/43 EEC) of coastal and low hill belts in a Mediterranean biodiversity hot spot (Gargano – Italy). Plant Biosyst. 147, 1006-1028.

Queiroz, C., Beilin, R., Folke, C., Lindborg, R., 2014. Farmland abandonment: threat or opportunity for biodiversity conservation? A global review. Front. Ecol. Environ. 12, 288-296.

Ramezani, H., Holm, S., Allard, A., Ståhla, G., 2013. A review of sampling-based approaches for estimating landscape metrics. Nor. J. Geogr. 67, 61-71.

Remmel, T.K., Fortin, M.-J., 2013. Categorical, class-focused map patterns: characterization and comparison. Landsc. Ecol. 28, 1587-1599.

Riiters, K.H., O'Neill, R.V., Jones, K.B., 1997. Assessing habitat suitability at multiple scales: a landscape-level approach. Biol. Conserv. 81, 191-202.

Rocchini, D., Perry, G.L.W., Salerno, M., Maccherini, S., Chiarucci, A., 2006. Landscape change and the dynamics of open formations in a natural reserve. Landsc. Urban Plan. 77, 167-177.

Russo, D., 2007. Effects of land abandonment on animal species in Europe: conservation and management implications. Integrated Assessment of vulnerable ecosystems under global change in the European Union. European Commission, Directorate – General for Research Environment, Office for Official Publications of the European Communities, Luxembourg.

Saura, S., Torras, O., Gil-Tena, A., Pascual-Hortal, L., 2008. Shape irregularity as an indicator of forest biodiversity and guidelines for metric selection, in: Lafortezza, R., Sanesi, G., Chen, J., Crow, T.R. (Eds.), Patterns and processes in forest landscapes. Springer, Dordrecht, pp. 167-189.

Schindler, S., von Wehrden, H., Poirazidis, K., Wrbka, T., Kati, V., 2013. Multiscale performance of landscape metrics as indicators of species richness of plants, insects and vertebrates. Ecol. Indic. 31, 41-48.

Schlossberg, S., King, D.I., 2009. Postlogging succession and habitat usage of shrubland birds. J. Wildl. Manag. 73, 226-231.

Símová, P., Gdulová, K., 2012. Landscape indices behavior: A review of scale effects. Appl. Geogr. 34, 385-394.

Sirami, C., Brontons, L., Martin, J.-L., 2007. Vegetation and songbird response to land abandonment: from landscape to census plot. Divers. Distrib. 13, 42-52.

Sitzia, T., Campagnaro, T., Gatti, E., Sommacal, M., Kotze, D.J., 2015. Wildlife conservation through forestry abandonment: responses of beetle communities to habitat change in the Eastern Alps. Eur. J. For. Res. 134, 511-524.

Sitzia, T., Dainese, M., Clementi, T., Mattedi, S., 2014. Capturing cross-scalar variation of habitat selction with grid sampling: an example with hazel grouse (*Tetrastes bonasia* L.). European Journal of Wildlife Research 60, 177-186.

Sitzia, T., Semenzato, P., Trentanovi, G., 2010. Natural reforestation is changing spatial patterns of rural mountain and hill landscapes: A global overview. For. Ecol. Manag. 259, 1354-1362.

Sitzia, T., Trentanovi, G., 2011. Maggengo meadow patches enclosed by forests in the Italian Alps: evidence of landscape legacy on plant diversity. Biodivers. Conserv. 20, 945-961.

Steckel, J., Westphal, C., Peters, M.K., Bellach, M., Rothernwoehrer, C., Erasmi, S., Scherber, C., Tscharntke, T., Steffan-Dewenter, I., 2014. Landscape composition and configuration differently affect trap-nesting bees, wasps and their antagonists. Biol. Conserv. 172, 56-64.

Stehman, S.V., 2012. Sampling strategies for forest monitoring from global to national levels, in: Achard, F., Hansen, M.C. (Eds.), Global forest monitoring from earth observation. CRC Press, Boca Raton, Florida, USA, pp. 79-105.

Susmel, L., 1958. Piano di riordinamento della proprietà silvo-pastorale Costantini in Val Tovanella (1958–1967). Azienda di Stato per le Foreste Demaniali, Ufficio di Belluno, Belluno, Italy.

Tasser, E., Walde, J., Tappeiner, U., Teutsch, A., Noggler, W., 2007. Land-use changes and natural reforestation in the Eastern Central Alps. Agric., Ecosyst. Environ. 118, 115-129.

Turner, M.G., Dale, V.H., Gardner, R.H., 1989a. Predicting across scales: theory development and testing. Landsc. Ecol. 3, 245-252.

Turner, M.G., Donato, D.C., Romme, W.H., 2013. Consequences of spatial heterogeneity for ecosystem services in changing forest landscapes: priorities for future research. Landsc. Ecol. 28, 1081-1097.

Turner, M.G., Gardner, R.H., O’Neill, R.V., 2001. Landscape ecology in theory and practice. Springer New York, New York, USA.

Turner, M.G., O’Neill, R.V., Gardner, R.H., Milne, B.T., 1989b. Effects of changing spatial scale on the analysis of landscape pattern. Landsc. Ecol. 3, 153-162.

Välimäki, P., Itämies, J., 2003. Migration of the clouded Apollo butterfly *Parnassius mnemosyne* in a network of suitable habitats – effects of patch characteristics. Ecography 26, 679-691.

Vaz, A.S., Marcos B., Gonçalves, J., Alves, P., Civantos, E., Lucas, R., Mairota, P., Garcia-Robles, J., Alonso, J., Blonda, P., Lomba, A., Honrado, J.P., 2015. Can we predict habitat quality from space? A multi-indicator assessment based on an automated knowledge-driven system. Int. J. Appl. Earth Obs. Geoinf. 37, 106-113.

Viola, F., Campedel, D., Toffolet, D., 2008. Introduction, in: Hardersen, S., Mason, F., Viola, F., Campedel, D., Lasen, C., Cassol, M. (Eds.), Research on the Natural Heritage of the Reserves Vincheto di Celarda and Val Tovanella (Belluno Province Italy), Conservation of two Protected Areas in the Context of a Life Project. Arti Grafiche Fiorini, Verona, pp. 17-24.

Wu, J., 2004. Effects of changing scale on landscape pattern analysis: scaling relations. Landsc. Ecol. 19, 125-138.

Wu, J., Buyantuyev, A., Jenerette, G.D., Litteral, J., Neil, K., Shen, W., 2011. Quantifying spatiotemporal patterns and ecological effects of urbanization: a multiscale landscape approach, Applied Urban Ecology. John Wiley & Sons, Ltd, pp. 33-53.

Wu, J., Jelinski, D.E., Luck, M., Tueller, P.T., 2000. Multiscale analysis of landscape heterogeneity: scale variance and pattern metrics. Geographic Information Sciences 6, 6-19.

Wu, J., Shen, W.J., Sun, W.Z., Tueller, P.T., 2002. Empirical patterns of the effects of changing scale on landscape metrics. Landsc. Ecol. 17, 761-782.

Zurlini, G., Girardin, P., 2008. Introduction to the special issue on "Ecological indicators at multiple scales". Ecol. Indic. 8, 781-782.

6. Paper V: Wildlife conservation through forestry abandonment: responses of beetle communities to habitat change in the Eastern Alps[5]

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Abstract

Research on changes in biodiversity due to the abandonment of forestry is important in understanding the role of reserves in conservation. The aim of this study was to investigate whether changes have occurred in species richness, abundance and composition of ground, longhorn and bark beetles due to habitat changes as a result of the cessation of forest management. We surveyed ten managed and ten abandoned forest plots in two watersheds located in the north-eastern Italian alpine region, which share a common history of use, climate regimes, stand structure and topography. Ground beetles, and longhorn and bark beetles were collected with pitfall and flight-intercept window traps, respectively, from May to mid-October 2010. The three beetle taxa responded differently to changes in habitat features and management cessation. Differences in individual species responses between the two watersheds may indicate a role of management abandonment through its impact on forest habitat structure. For instance, ground beetle species mainly responded negatively to soil moisture and positively to understorey vegetation cover. Unexpectedly, saproxylic species responded variably, and often negatively, to deadwood features in these forests, but did respond positively to the volume of standing *Abies alba* trees. The assemblages of carabids and bark beetles differed between the two watersheds. Our results confirmed that 50 years of forest management cessation resulted in changes in the biodiversity of beetles in alpine forests, likely due to their response to changes in habitat structure. Moreover, we expect that where the unplanned abandonment of forestry practices and habitat rewilding are undergoing, like in many marginal areas of Europe, similar habitat structure dynamics and beetle responses are likely to occur spontaneously.

Keywords: Biodiversity conservation, Forest management, Old-growth forests, Carabidae, Cerambycidae, Scolytinae

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6.1 Introduction

Set-asides in forest reserves enable the development of natural dynamics and initiate a rewilding in forest structure over time. Therefore, studies investigating changes in biodiversity due to structural changes in forests as a result of the abandonment of forestry are important for understanding the role of reserves in conservation. For example, after the abandonment of management, the volume of dead wood (Green and Peterken 1997; Ódor and Standovár 2001; Vuidot et al. 2011), the diversity of tree species (Schmidt 2005; Sitzia et al. 2012) and the number of microhabitats increase (Winter and Möller 2008; Larrieu et al. 2012). As previously managed forests resume successional changes over time, so does biodiversity. However, changes towards more natural conditions may be slow and not observable in the short term (Brunet et al. 2010; Paillet et al. 2010). In general, the abandonment of forestry practices is thought to be beneficial for forest specialists dependent on old-growth forest characteristics (Bengtsson et al. 2000), yet abandonment may also result in habitat homogenisation and the loss of species favoured by different disturbance regimes (see Taboada et al. 2006, 2008).

Invertebrates are an ideal group to study the effects of a changing forest structure brought about by forest management abandonment, as they are responsive to anthropogenic disturbances and environmental change (Desender et al. 1991). Invertebrates are also essential in most ecosystems due to their functional role in many processes, such as decomposition, pollination, predation, trophic interactions, and as prey for many other taxa (Samways 2007). Among invertebrates, ground (*Carabidae*), longhorn (*Cerambycidae*) and bark (*Scolytinae*) beetles have been extensively studied and their ecology is well known (e.g. Linsley 1959; Rudinsky 1962; Kotze et al. 2011). Furthermore, longhorn and bark beetles have received increasing attention as they are saproxylic, i.e. they are dependent during some part of their life cycle on deadwood (see Siitonen 2001; Grove 2002). Several studies have dealt with the effects of changing forest structure on these beetle groups and showed that particular species or subsets of species will respond differently and to different extents. For example, forestry practices have varying effects on ground beetles (Similä et al. 2003; Niemelä et al. 2007) and saproxylic beetles respond discordantly depending on deadwood characteristics (Similä et al. 2003; Hjältén et al. 2007). Deadwood accumulates in managed forests when definite silvicultural precautions are applied or when forestry exploitation ceases and the natural unexploited wild state is restored. Some authors have shown a positive effect of management abandonment and less intensive forestry on their biodiversity (e.g. Niemelä et al. 2007; Taboada et al. 2008). In particular, saproxylic beetles most likely benefit from forest abandonment due to an increase in dead and decaying wood as a result of forest succession (e.g. Müller et al. 2008, 2010; Bouget et al. 2014).

In Europe, knowledge regarding the effects of forest management and forest habitat attributes on beetle communities primarily derives from Fennoscandian countries, highlighting the need for research elsewhere, such as in alpine areas (but see Lemperiere and Marage 2010). Furthermore, the lack of studies on the abandonment of forestry in alpine regions is related to the difficulty in controlling confounding factors due to the high variability of forests (Sitzia et al. 2012), and to the lack of undisturbed forests per se (Motta et al. 2006).

Our study investigates changes in ground, longhorn and bark beetle communities in relation to forest habitat features at the stand level in two differently managed areas. First, we evaluated beetle diversity, expecting that habitat features shaped by management abandonment will enable forests to host an increased diversity in saproxylic beetles, but a lower carabid beetle diversity (due to the loss of generalist and open-habitat carabid species, see Mullen et al. 2008). Second, we tested the responses of individual species and communities to habitat features, predicting that species associated with habitat features similar to those of undisturbed mature forests, such as forest specialists, will increase in abundance. Finally, we discuss the conservation implications of changes in habitat features and forest management abandonment, given the responses of ground, longhorn and bark beetle communities.

6.2 Materials and Methods

6.2.1 Study area

The study was conducted in the contiguous Tovanella and Cajada watersheds (1040 ha each) located in the alpine area of north-eastern Italy. Both areas are extensively covered by forests and were intensively managed between 1943 and 1953. In Tovanella, silvicultural and grazing activities ceased in 1957 (Susmel 1958), while in Cajada, forests are non-intensively managed, i.e. with a group selection system for the production of timber (Andrich 2005). Tovanella and Cajada are in close proximity, sharing a similar altitudinal range, climate and substrate (Sitzia et al. 2012), with no substantial biogeographic differences between the two areas. Both forested watersheds fall within Natura 2000 sites and the studied forests were classified as habitats of community interest. Information regarding general characteristics of these areas is presented in Table 1 .We recognise that our study design does not include true replication and that the patterns observed may not be solely due to the cessation of forest management. Our focus is, therefore, on reporting differences in habitat features to which individual species and the beetle communities respond.

6.2.2 Habitat survey

Ten circular plots with a radius of 12.5 m were randomly placed in silver fir (*Abies alba*) forests of both areas, avoiding forest edges and slopes steeper than 26°. Minimum distance between plots was 200 m. Understory vascular flora was surveyed, and species cover was assigned using the Braun–Blanquet (1932) index of cover. For all trees (DBH>7.5 cm), we recorded DBH, species, total height, height of crown insertion and four vertical crown projection radii. The volume of each tree within the plot was calculated and then transformed to hectares applying different species-specific two-way volume tables used in the Italian National Inventory (Castellani et al. 1984). For different types (log, snag, stump) of coarse woody debris (CWD), several metrics were collected in order to calculate the total volume.

Table 1: Study area characteristics

	Tovanella	Cajada
Study area characteristics		
Mean annual temperature (°C)	7.2	
Mean annual precipitation (mm $year^{-1}$)	1300-1500	
Precipitation peaks	May–June and October–November	
Altitudinal range (m a.s.l.)	550-2500	
Watershed characteristics		
Altitude (m a.s.l.)	1221 ± 103	1228 ± 43
Slope (°)	14.8 ± 8.0	16.1 ± 5.4
Basal Area (m^2 ha^{-1})	46.8 ± 10.0	50.5 ± 13.2
Mean annual harvest (m^3 ha^{-1})	-	1.72
Syntaxonomical unit	*Adenostylo glabrae-Abietetum albae* H. Mayer et A. Hofmann 1969	
Habitat Natura 2000	9130 *Asperulo-Fagetum* beech forests	
Main tree species	*Abies alba* Mill., *Picea abies* (L.) H. Karst., *Fagus sylvatica* L.	

A single value is given when the two areas share common characteristics

Naturally occurring logs were sampled by applying the line intersect method and volume was calculated accordingly (Van Wagner 1982; Marshall et al. 2000), whereas stumps and snags were sampled in two transects (50 m X 8 m) with their mid-point at the centre of the 12.5 m plot. Unitary volume of stumps was calculated applying the truncated cone volume formula, and unitary volume of snags was computed by applying species- specific two-way volume tables. For each piece of CWD, a decay status was assigned by dividing the decay stages into four and five classes for stumps (Motta et al. 2006) and for snags and logs (Maser et al. 1979; Sollins 1982), respectively. A decay-weighted value of stumps, snags and logs was computed by weighing the volume of each deadwood piece by its decay class. Plant species intersecting the central line of the above-mentioned two transects at every 1-m segment were recorded to derive Ellenberg indicator values for soil moisture, reaction and nitrogen (Ellenberg et al. 1991). These values were a community-level weighted mean (Lavorel et al. 2008) for each transect using frequency data of species composition. Further information on the sampling design and data collection is available in Sitzia et al. (2012).

6.2.3 Beetle sampling

Two different types of traps were used to sample beetles within the 12.5-m-radius plots. Ground beetles were collected using two pitfall traps per plot, placed 5–7 m apart, whereas longhorn and bark beetles were sampled using one flight-intercept window trap that was placed hanging from a tree close to the centre of each plot. Pitfall traps were plastic cups (7 cm diameter, 12 cm depth) buried in the ground with their tops flush with the ground level, half-filled with a vinegar solution. Individuals collected in the two pitfall traps at the same plot were pooled together per visit. Window traps consisted of orthogonal panels of transparent plexiglas (60 cm X 40 cm) that were positioned perpendicular to the ground level and with a funnel positioned underneath it. The intercepted beetles were collected in a bottle half-filled with ethanol (0.5 l total capacity) that was located under the traps at a height of 1–1.8 m depending on branch availability. Both pitfall and flight-intercept window traps were emptied every fortnight. Continuous sampling started at the beginning of May and lasted until mid-October 2010. The beetles were collected eight and nine times in Cajada and Tovanella, respectively. All collected individuals were sorted and identified to species level (see Supplementary Material Table S1 for identification keys used).

6.2.4 Statistical analysis

To test our hypotheses, the following analyses were performed. First, to have a general overview, we simply compared species richness between the two watersheds (Tovanella and Cajada) by rarefying species richness of the total catch per taxonomic group.

Second, generalised linear mixed models (GLMMs) were run to evaluate the responses of the total number of individuals per taxonomic group, and of individually collected abundant species (of at least 45 individuals in total) to a suite of habitat variables by taking into consideration their location. Eight ground and five bark beetle species could be analysed individually, while the low-abundance species were pooled into groups based on their a priori predicted responses. None of the longhorn beetles were collected in sufficient number to be analysed individually (the most abundant longhorn beetle in our dataset, *Rhagium mordax*, was represented by only ten individuals). Overall number of individuals and the number of individuals of the abundant species and the low-abundance groups were modelled following a Poisson distribution (see O'Hara and Kotze 2010), and overdispersion was accounted for by fitting an observation-level random term. Study plot nested within watershed (Tovanella, Cajada) was added as a random factor to account for possible spatial autocorrelation in the design. Collecting visit was added to the models as a fixed factor to reflect the time of the season, and the log number of trapping days was added as an 'offset term' to account for differences in trapping days in the field (Kotze et al. 2012). Predictor variables for ground and saproxylic beetles are presented in Table 2. Variables used in the same model did not correlate strongly with one another (maximum $r < 0.7$). Models were simplified by removing non-significant (here $p > 0.1$) terms one at a time until only significant variables remained. However, for ground beetles, soil moisture, soil reaction, understorey vegetation and crown cover were kept in the models even if non-significant due to their reported importance in the literature. For saproxylic beetles, the decay-weighted stumps, snags and logs were always retained in the models.

All habitat variables in the ground, longhorn and bark beetle models were standardised to zero mean and unit variance to evaluate their relative contributions to the beetle response (Schielzeth 2010).

Finally, non-metric multidimensional scaling (NMDS) was used to investigate the assemblage structure of the beetles in relation to habitat features that were collected from the different plots in the two forested watersheds (Tovanella and Cajada). We performed one NMDS per taxon using the Bray–Curtis coefficient as the dissimilarity measure and permutation tests in the vector-fitting procedure. Habitat variables included in the NMDSs are presented in Table 2. All analyses were performed in the R statistical software, version 3.0.2 (R Development Core Team 2013), using the 'rarefy' and 'metaMDS' functions in the 'vegan' library (Oksanen et al. 2013) and the 'glmer' function in the 'lme4' library (Bates et al. 2014).

Table 2: Habitat variables used in the GLMMs and NMDSs.

Habitat variables	GLMM	NMDS	Tovanella	Cajada
Soil moisture (Ellenberg values)	G	GS	4.9 ± 0.1	5.0 ± 0.3
Soil reaction (Ellenberg values)	G	GS	5.0 ± 0.5	5.0 ± 0.8
Soil nitrogen content (Ellenberg values)		S	5.6 ± 0.4	4.6 ± 0.7
Understorey vegetation (%)	G	GS	72.7 ± 13.1	78.3 ± 18.3
Decay-weighted stumps (m^3 ha^{-1})	GS	GS	19.5 ± 11.6	104.5 ± 101.4
Decay-weighted snags (m^3 ha^{-1})	GS	GS	31.8 ± 21.0	6.7 ± 16.5
Decay-weighted logs (m^3 ha^{-1})	GS	GS	50.7 ± 31.7	0.2 ± 0.2
Total tree volume (m^3 ha^{-1})	G	G	556±154	663 ± 183
Volume of *Abies alba* (m^3 ha^{-1})	S	S	278 ± 144	442 ± 197
Volume of *Picea abies* (m^3 ha^{-1})	S	S	193 ± 89	197 ± 162
Volume of *Fagus sylvatica* (m^3 ha^{-1})	S	S	61 ± 41	24 ± 25
Total crown cover (%)	GS	GS	141.9 ± 26.3	106.7 ± 13.7
Tree species richness (no.)	S	S	4.2 ± 0.9	2.9 ± 0.3

Letters indicate which variables were used for the beetle groups: ground beetles (G), longhorn and bark beetles (S), and all three groups (GS). Mean values and standard deviations are given for the Tovanella and Cajada watershed forests (see Sitzia et al. 2012).

6.3 Results

We collected a total of 19 ground beetle, 15 longhorn beetle and 17 bark beetle species (Supplementary Material, Table S2). A higher rarefied number of carabid species were collected from the managed compared with the abandoned watershed (Cajada = 18.0, Tovanella = 12.8, n = 1682 individuals). No difference existed in the rarefied number of longhorn (Cajada = 6.0, Tovanella = 5.7, n = 8 individuals) and bark beetle (Cajada = 6.0, Tovanella = 6.0, n = 166 individuals) species collected from these watersheds. A total of 4341 ground beetle, 47 longhorn beetle and 1515 bark beetle individuals were collected, with significantly more ground and bark beetles collected from the Tovanella than from the Cajada watershed (Tables 3, 4; Fig. 1). Longhorn beetle abundances did not differ between the watersheds.

Increasing soil moisture affected all analysed carabid species negatively (except for *Carabus creutzeri kircheri*, yet insignificantly so), while an increase in understorey vegetation cover affected all ground beetle species positively. The effects of increasing soil reaction and crown cover were variable, but mainly positive (Table 3; Fig. 2). The decay-weighted volumes of snags, stumps and logs were occasionally important to some carabid beetle species. Interestingly, the effects of these habitat features were always negative, meaning that an increase in CWD had a negative effect on the number of individuals of these species (Table 3). Longhorn and bark beetles responded inconsistently to deadwood, yet for stumps and logs the effects were mainly, and unexpectedly, negative. Few species responded statistically significantly to CWD; *Hylastes cunicularius* responded positively to snags, while *Dryocoetes autographus* responded negatively to logs (Table 4; Fig. 3). Three bark beetle species responded significantly and positively to the volume of *A. alba* trees (*H. cunicularius*, *Trypodendron domesticum*, *D. autographus*), while *T. domesticum* also responded negatively to the volume of *Picea abies* trees (Table 4). Most species were more abundant in forests of the Tovanella than of the Cajada watershed, many significantly so (Tables 3, 4; Fig. 1). None of the carabid beetle species analysed were significantly more abundant in Cajada, the watershed that is still under management.

NMDS analysis revealed that the composition of the ground beetle assemblage was marginally influenced by soil moisture ($r^2 = 0.324$, $p = 0.090$) and crown cover ($r^2 = 0.328$, $p = 0.095$), and significantly by the decay-weighted logs ($r^2 = 0.403$, $p = 0.037$) (Fig. 4 a). Among carabids, only *Pterostichus quadrifoveolatus* and *Synuchus vivalis vivalis* responded strongly to the habitat variables, both being positively associated with increasing soil moisture (Fig. 4 b). For longhorn beetles, decay-weighted logs ($r^2 = 0.445$, $p = 0.024$) and the number of trees ($r^2 = 0.495$, $p = 0.017$) were significantly associated with their assemblage, both of which showed higher values in Tovanella than in Cajada (Fig. 5 a).

Table 3: Ground beetle generalized linear mixed model results.

	Int	Watersh.	Moist	React	Veg	CC	TTV	Snag	Stump	Log
All carabid individuals	0.197	0.601	-0.452	0.146	0.472	0.098		-0.187		
	(0.161)	(0.233)	(0.093)	(0.105)	(0.107)	(0.100)		(0.096)		
	0.223	**0.010**	**<0.001**	0.165	**<0.001**	0.327		**0.051**		
Abax parallelepipedus	-3.299 (0.321)	0.512	-0.281	0.145	0.831	0.125				
	<0.001	(0.444)	(0.196)	(0.188)	(0.230)	(0.206)				
		0.250	0.152	0.441	**<0.001**	0.543				
Molops piceus austriacus	-3.980 (0.488)	2.131	-0.203	0.112	0.149	0.100				
	<0.001	(0.699)	(0.341)	(0.328)	(0.369)	(0.324)				
		0.002	0.551	0.733	0.686	0.757				
Pterostichus burmeisteri	-0.699 (0.220)	0.436	-0.398	0.127	0.304	0.061		-0.268		
	0.001	(0.297)	(0.119)	(0.136)	(0.137)	(0.127)		(0.125)		
		0.142	**<0.001**	0.350	**0.02**	0.634		**0.032**		
Abax pilleri	-2.333 (0.315)	1.131	-0.878	0.219	0.957	0.048				
	<0.001	(0.438)	(0.242)	(0.211)	(0.247)	(0.211)				
		0.010	**<0.001**	0.300	**<0.001**	0.818				
Notiophilus biguttatus	-3.462 (0.475)	-0.976	-0.054	-0.094	0.332	0.104				
	<0.001	(0.702)	(0.251)	(0.287)	(0.316)	(0.346)				
		0.165	0.830	0.742	0.293	0.763				
Group forest	-3.794 (0.529)	-1.193	-0.829	-0.121	0.683	-0.327				
	<0.001	(0.713)	(0.346)	(0.369)	(0.463)	(0.369)				
		0.095	**0.017**	0.761	0.140	0.376				
Carabus creutzeri kircheri	-4.140 (0.364)	2.328	0.086	0.641	0.499	-0.007	-0.333			
	<0.001	(0.414)	(0.248)	(0.137)	(0.206)	(0.127)	(0.176)			

	Int	Watersh.	Moist	React	Veg	CC	TTV	Snag	Stump	Log
		<0.001	0.728	**<0.001**	**0.016**	0.954	**0.058**			
Carabus linnaei	-2.669 (0.247)	1.110	-0.730	0.045	0.486	0.254	0.456		-0.431	-0.313
	<0.001	(0.356)	(0.159)	(0.138)	(0.142)	(0.131)	(0.142)		(0.157)	(0.169)
		0.002	**<0.001**	0.745	**<0.001**	**0.053**	**0.001**		**0.006**	**0.073**
Pterostichus unctulatus	-2.254 (0.611)	-0.378	-1.642	0.770	1.822	0.497				
	<0.001	(0.935)	(0.570)	(0.457)	(0.582)	(0.463)				
		0.686	**0.004**	0.092	**0.002**	0.283				
Group Generalists	-3.944 (0.561)	-1.293	-0.346	-0.847	0.040	-0.376				
	<0.001	(0.850)	(0.271)	(0.463)	(0.498)	(0.399)				
		0.128	0.202	0.068	0.935	0.346				

Species and species groups are listed *a priori* from forest associated (top) to generalist and open habitat species (bottom). Values per species represent coefficients, standard error (in brackets) and *p*-values. Significant *p*-values are indicated in bold face. Int = intercept; Watersh. = the Cajada (which is in the intercept) and Tovanella watersheds; Moist = soil moisture; React = soil reaction; Veg = understory vegetation %; CC = crown cover %; TTV = total tree volume; Snag = decay-weighted snags, Stump = decay-weighted stumps, Log = decay-weighted logs.

Table 4: Longhorn and bark beetle generalized linear mixed model results.

	Int	Watersh.	Vol-*Abies*	Vol-*Picea*	Vol-*Fagus*	CC	NoT	Snag	Stump	Log
Cerambycidae										
All cerambycid individuals	-4.259 (0.463)	1.014						-0.071 (0.190)	-0.314 (0.429)	0.212 (0.184)
		(0.681)								
	<0.001	0.136						0.708	0.464	0.250
Scolytinae										
All scolytid individuals	-1.625 (0.338)	2.700					-0.410 (0.162)	0.029 (0.139)	-0.130 (0.158)	-0.168 (0.187)
		(0.580)								
	<0.001						**0.011**	0.836	0.410	0.369

	Int	Watersh.	Vol-*Abies*	Vol-*Picea*	Vol-*Fagus*	CC	NoT	Snag	Stump	Log
		<0.001								
Xylosandrus germanus	-5.136 (0.693) <0.001	3.750 (1.175) **0.001**						-0.217 (0.364) 0.551	-0.483 (0.612) 0.431	-0.063 (0.407) 0.876
Hylastes cunicularius	-1.417 (0.253) <0.001	0.659 (0.438) 0.133	0.363 (0.120) **0.003**			0.426 (0.141) **0.003**		0.460 (0.104) **<0.001**	0.163 (0.135) 0.227	-0.036 (0.160) 0.825
Trypodendron domesticum	-6.288 (0.755) <0.001	3.761 (1.162) **0.001**	0.546 (0.237) **0.021**	-0.577 (0.199) **0.004**		0.576 (0.207) **0.005**	-0.711 (0.256) **0.006**	0.215 (0.156) 0.168	-0.792 (0.752) 0.292	-0.092 (0.200) 0.645
Xyleborus dispar	-3.528 (0.510) <0.001	1.530 (0.804) 0.057						-0.006 (0.273) 0.981	-0.215 (0.345) 0.534	-0.580 (0.359) 0.106
Dryocoetes autographus	-5.276 (0.743) <0.001	2.829 (1.200) **0.018**	0.694 (0.320) **0.030**					-0.232 (0.351) 0.508	-0.319 (0.697) 0.648	-0.898 (0.429) **0.036**
Low-abundance scolytid species	-2.690 (0.424) <0.001	0.528 (0.673) **0.433**						-0.183 (0.220) 0.405	-0.211 (0.244) 0.389	0.011 (0.255) 0.965

See Table 3 for details. Int = intercept; Watersh. = the Cajada (which is in the intercept) and Tovanella watersheds; Vol-*Abies* = Volume of *Abies alba*, Vol-*Picea* = Volume of *Picea abies*, Vol-*Fagus* = Volume of *Fagus sylvatica*; CC = crown cover %; NoT = tree species richness; Snag = decay-weighted snags, Stump = decay-weighted stumps, Log = decay-weighted logs.

Apart from the clear separation of the two watersheds in terms of the bark beetle assemblage, several variables had a significant effect on these beetles: crown cover (r^2 = 0.595, p = 0.002), decay-weighted logs (r^2 = 0.601, p = 0.006), decay-weighted snags (r^2 = 0.434, p = 0.032), soil nitrogen (r^2 = 0.471, p = 0.022), and marginally for tree species richness (r^2 = 0.347, p = 0.089) and decay-weighted stumps (r^2 = 0.345, p = 0.077) (Fig. 6 a). *Scolytus intricatus* and *T. domesticum* were associated with decay-weighted snags and logs, as well as with tree species richness and crown cover, while *Cryphalus abietis* was positively associated with the volume of *A. alba* trees (Fig. 6 b). The composition of ground and bark beetle assemblages (r^2 = 0.159, p = 0.044 and r^2 = 0.482, p < 0.001, respectively), but not longhorn beetle assemblages (r^2 = 0.070, p = 0.298), was significantly different between the two watersheds (Figs. 4, 5, 6).

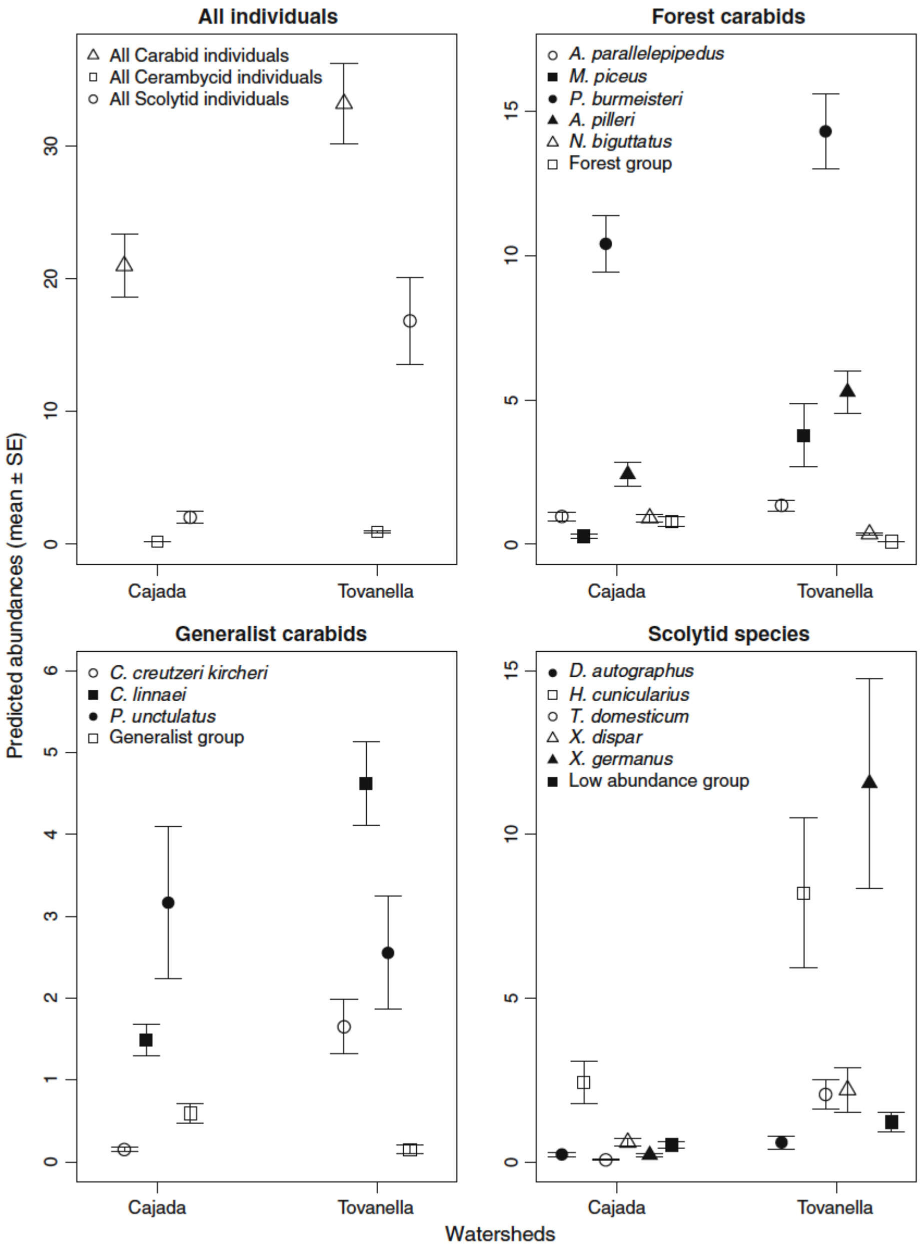

Figure 1: Predicted beetle abundances (means ± SE) in the two forested watersheds, Cajada and Tovanella. Four plots are presented, including all individuals, forest carabid beetles, generalist carabid beetles and bark beetles. Predicted values were calculated using the 'predict' function in R, given all other variables in the model, and on averaging over all possible values of the random effect (e.g., marginalised predictions).

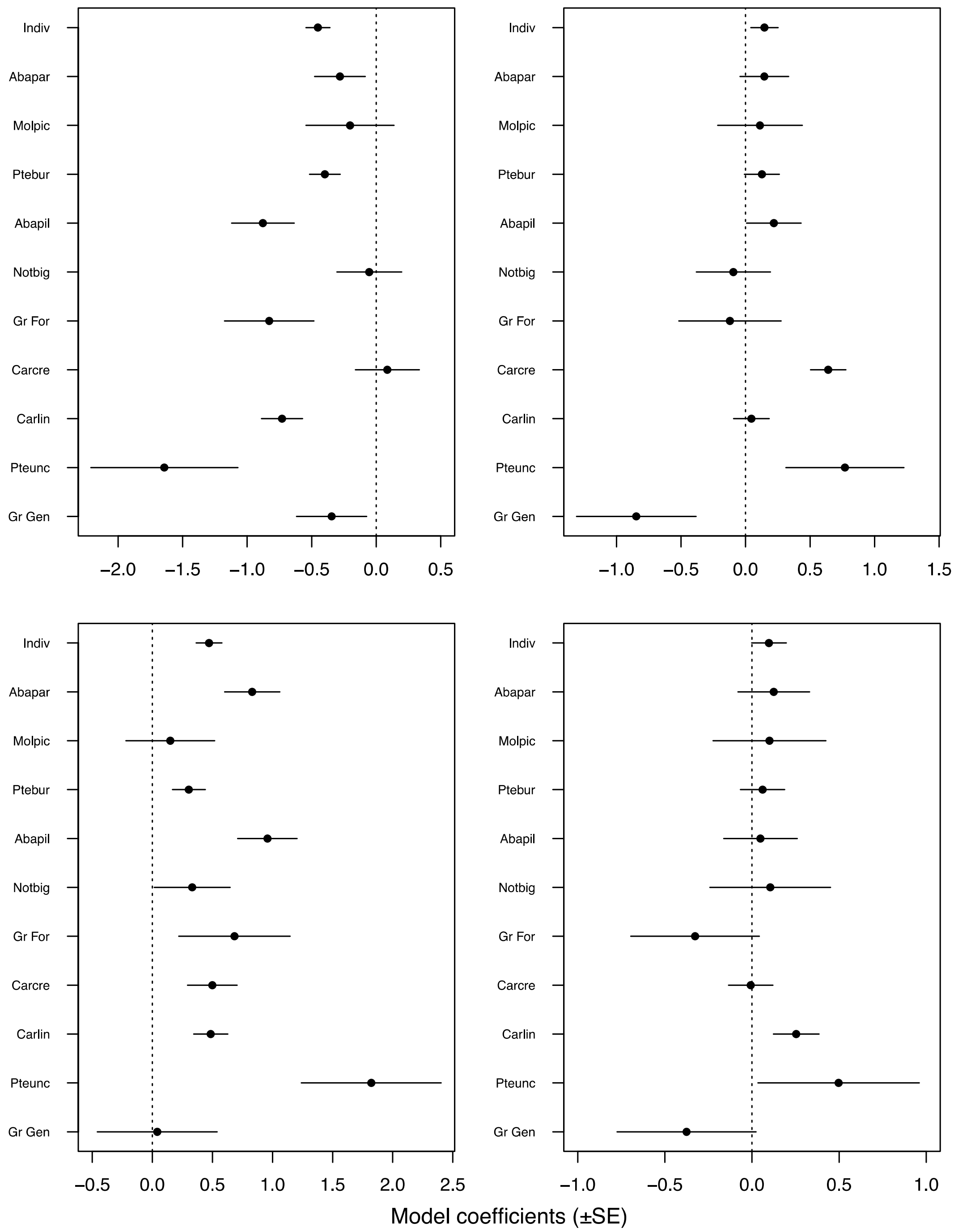

Figure 2: Ground beetle responses (model coefficients ± 1SE, see Table 3) to soil moisture, soil reaction, % understorey vegetation and % crown cover. Species and species groups are listed *a priori* from forest associated (*top*) to generalist and open habitat species (*bottom*). Species abbreviations consist of the *first three letters* of the genus and species name. For example, Abapar = *Abax parallelepipedus*. Indiv = All carabid beetle individuals. Gr For = group of low abundance forest species. Gr Gen = list of low abundance generalist species.

6.4 Discussion

Habitat features affected the beetle assemblages within the two forested watersheds.

While ground and bark beetle assemblages often responded to certain habitat features, small sample size prevented us from fully evaluating longhorn beetle responses. Nevertheless, in general, beetle responses to habitat features enabled us to better understand how structurally different forests can influence beetle assemblages.

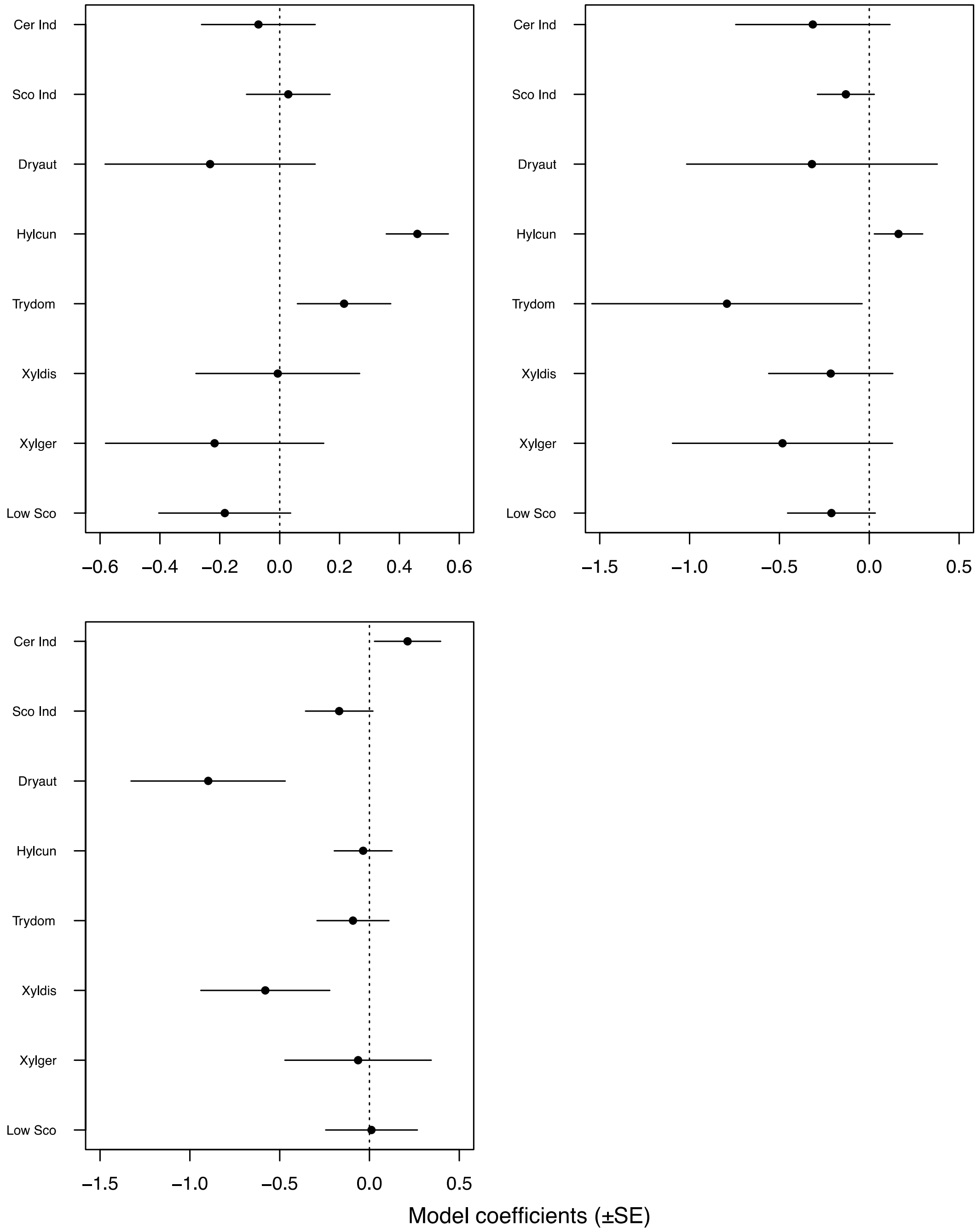

Figure 3: Longhorn and bark beetle responses (model coefficients ± 1SE, see Table 4) to the decay-weighted snags, stumps and logs. Species abbreviations consist of the *first three letters* of the genus and species name. For example, Dryaut = *Dryocoetes autographus*. Cer Ind = All longhorn beetle individuals, Sco Ind = all bark beetle individuals, Low Sco = group of low abundance bark beetle species.

Carabid beetle communities and species responded mainly to soil moisture, understorey vegetation and deadwood. Several carabid species and species groups responded negatively to increasing soil moisture, but positively to increasing understorey vegetation cover. Saproxylic species and communities responded mainly to naturally occurring deadwood, crown cover, tree species richness and volumes. These results showed that at the plot level, several habitat features shape beetle assemblages in silver fir (*A. alba*) forests.

Forest management shapes tree cover, richness and composition, and its cessation favours the accumulation of naturally occurring deadwood. Changes in these habitat features played a role in shaping beetle assemblages and are at least partly caused by management cessation. Therefore, management cessation viewed as the sum of habitat feature changes can be a factor influencing beetle species and communities.

6.4.1 Habitat features

The three beetle taxa responded variably to the evaluated habitat variables. The response of ground beetles to moisture is well established (Thiele 1977; Niemelä et al. 1992). Moisture had a negative effect on almost all ground beetle species and species groups analysed. Silver fir forests in the Southern Alps are usually located on shaded and moist slopes (Mayer 1974), which may affect ground beetle adults, and/or their larvae negatively. All species and the community responded positively to an increase in the cover of understorey vegetation (e.g. Taboada et al. 2008, 2010) and to some degree to tree cover (e.g. Niemelä et al. 1996; Humphrey et al. 1999). We showed that increasing CWD negatively affected some carabid species, such as the wingless *Pterostichus burmeisteri* and *Carabus linnaei* which may have resulted from CWD hampering beetle movements on the forest floor (Sroka and Finch 2006) or affecting their catch rate. Furthermore, other invertebrates may take advantage of higher amounts of CWD. For example, deadwood is important for generalist predator spiders (Varady-Szabo and Buddle 2006) and ants. At the assemblage level, however, decay-weighted logs appeared to have an effect on ground beetles, supporting Cobb et al. (2007) and Fuller et al. (2008).

The varying albeit generally weak responses of bark beetle species to deadwood volumes of snags, logs and stumps, weighted by their decay stages, reinforces the importance of considering different deadwood types when investigating biodiversity (Similä et al. 2003; Lassauce et al. 2011). At the community level, however, responses to deadwood were stronger. Higher values for logs were associated with the abandoned watershed (Tovanella) and its associated longhorn and bark beetle communities, while also snags were positively related to the bark beetle community at Tovanella. Moreover, stumps—indicating relatively recent forestry operations—were positively related to the bark beetle community in the Cajada watershed. Contrasting results are found in the literature on saproxylic beetles. Composition was either highly similar in snags and stumps (Hedgren 2007), or differed between stumps and logs (Jonsell and Hansson 2011; Brin et al. 2013), logs and snags (Ulyshen and Hanula 2009; Bouget et al. 2012), or between all three deadwood types (Abrahamsson and Lindbladh 2006; Hjältén et al. 2010). However, all these studies, except for Bouget et al. (2012), are based on experiments in which deadwood was manipulated. Furthermore, in our case, abundance values enabled for a more in-depth analysis of the response to deadwood types, highlighting species-specific responses. However, only two species responded to even one of these features: *H. cunicularius* positively to snags, representing its premium feeding substrate, and *D. autographus*, which was negatively associated with logs. The latter response is not in agreement with the moist substrates of Norway spruce, which the low-intensity cutting system should provide. However, a decrease in the frequency of *D. autographus* at very large log sections has recently been observed (Kula et al. 2011). Moreover, this species requires that the phloem is not strongly degraded (Kacprzyk and Bednarz 2014), a condition which large and highly decayed logs do not meet adequately.

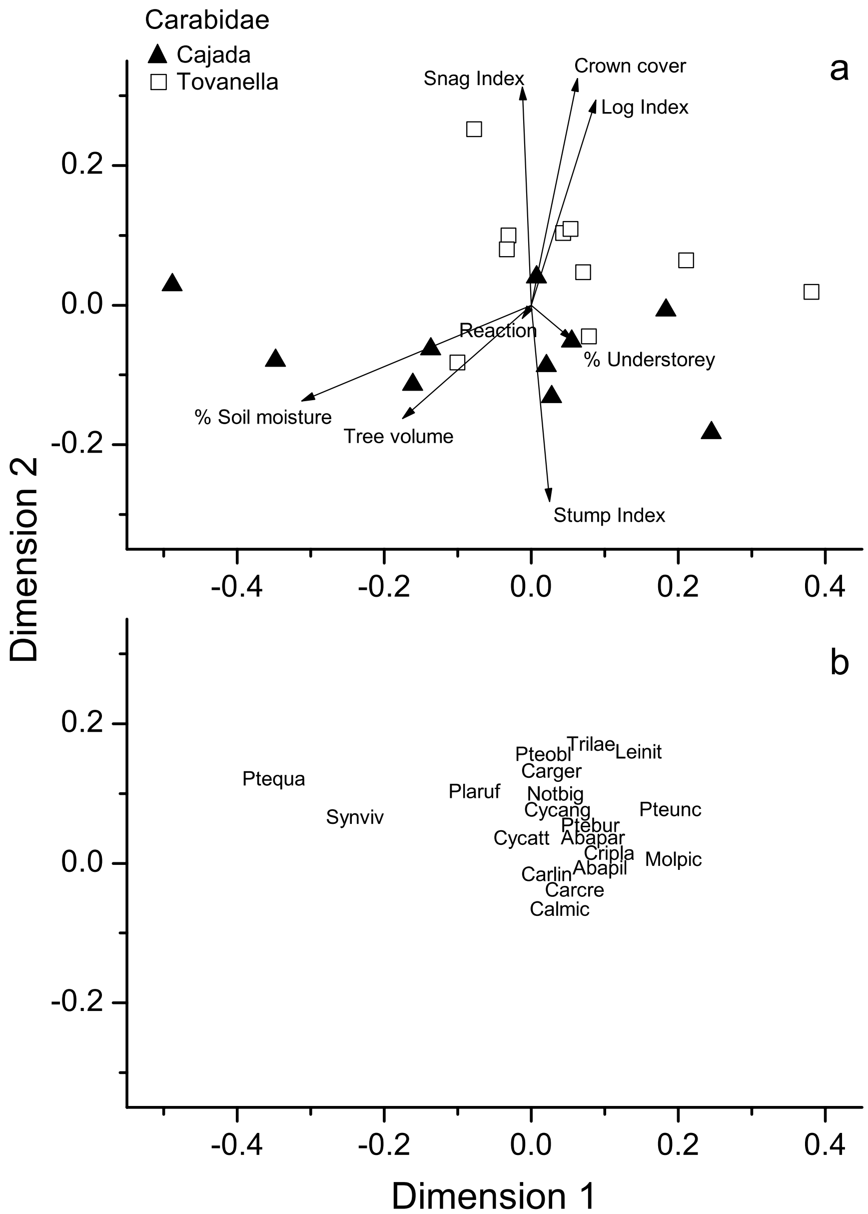

Figure 4: a) NMDS ordination of the ground beetle assemblage in the managed (*filled triangles*) and abandoned (*open squares*) forests. Two-dimensional stress value = 0.11. b) Ground beetle species ordination plot. For full species names, see the Supplementary Material, Table S2.

Bark beetles, many of which require specific host tree species (Rudinsky 1962), responded variably to the volumes of the different dominant tree species in these forests. Three of the five bark beetle species analysed (*H. cunicularius*, *T. domesticum*, *D.*

autographus) responded positively to the volume of silver fir, *A. alba*, while one (*T. domesticum*) also responded negatively to the volumes of spruce, *P. abies*. One possible explanation for the variation in our results may be related to the long history of exploitation that has influenced the species composition and amount of deadwood of certain tree species. Deadwood factors not investigated here may influence the response of saproxylic beetle communities and species.

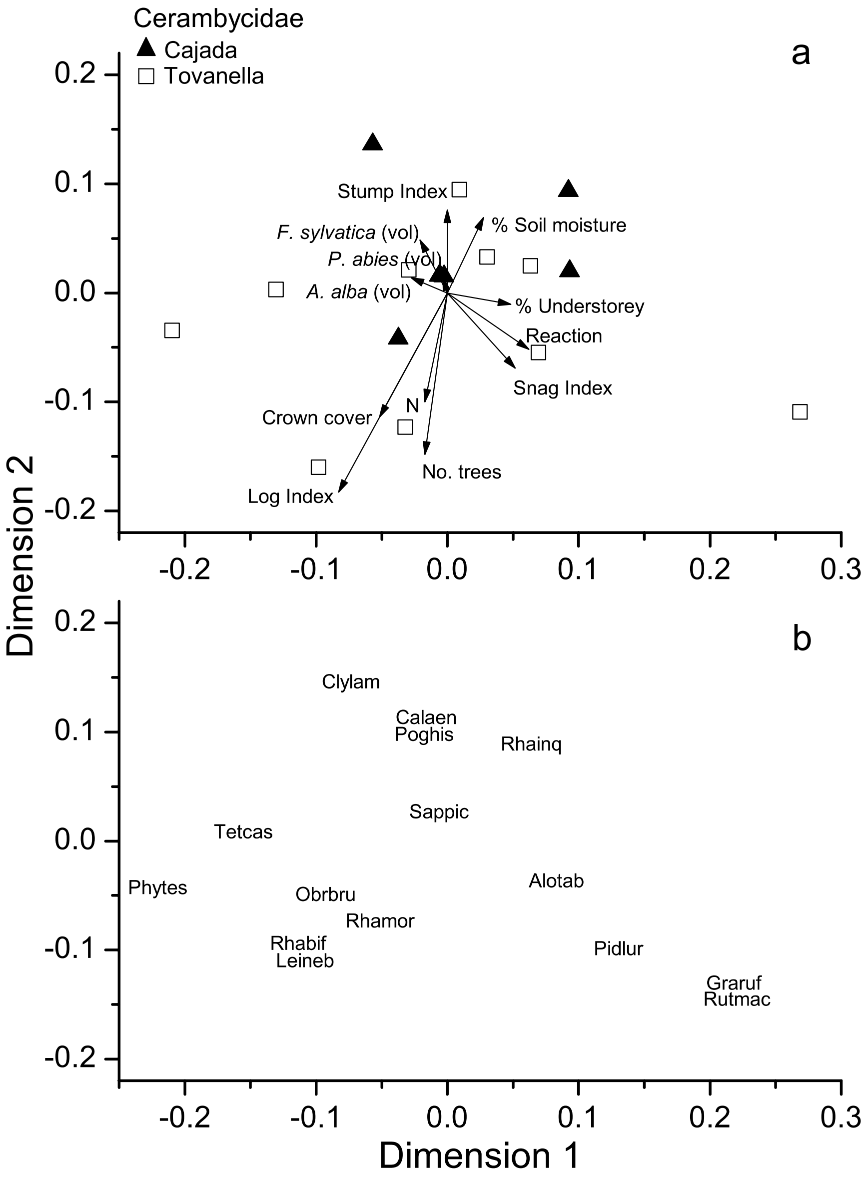

Figure 5: a) NMDS ordination of the longhorn beetle assemblage in the managed (*filled triangles*) and abandoned (*open squares*) forests. Two-dimensional stress value = 0.15. The two *filled triangles* at the centre of the ordination represent six of the 10 plots. b) Longhorn beetle species ordination plot. For full species names, see the Supplementary Material, Table S2.

Deadwood size and diversity are important features that affect saproxylic beetle diversity (Bouget et al. 2013, 2014). For example, large snags are richer in species than small ones (Bouget et al. 2012). Furthermore, heterogeneous microhabitat, moisture and fungal colonisation in different parts of living or dead trees may help in explaining the occurrence of certain saproxylic species (Jonsell et al. 2005; Abrahamsson and Lindbladh 2006; Winter and Möller 2008).

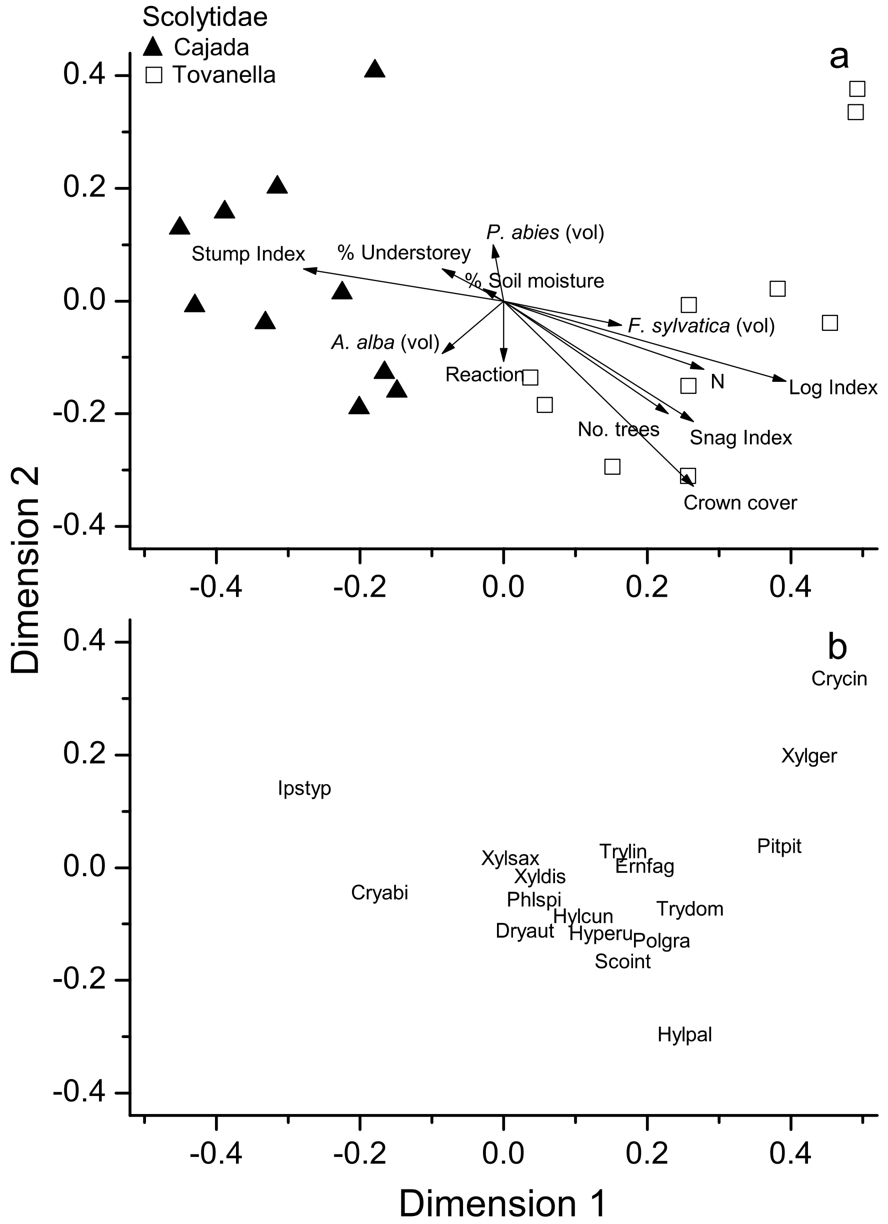

Figure 6: a) NMDS ordination of the bark beetle assemblage in the managed (*filled triangles*) and abandoned (*open squares*) forests. Two-dimensional stress value = 0.09. b) Bark beetle species ordination plot. For full species names, see the Supplementary Material, Table S2.

6.4.2 Forest management cessation

Forest management cessation is generally considered a successful *non-action* to maintain biodiversity by enabling the rewilding of ecological processes. In the Tovanella watershed, forestry practices were abandoned more than 50 years ago, while forest management still continues in the Cajada watershed. These different management trajectories have caused differences in habitat features between the plots located in the two forested watersheds (Sitzia et al. 2012). Many of these habitat features influenced beetle composition and abundance. Volumes of naturally occurring logs and snags were clearly higher in abandoned than in managed plots. These habitat features influenced the composition of ground, bark and longhorn beetles and influenced the abundance of several ground and bark beetles. Furthermore, management cessation is positively related to crown cover that in turn had a positive influence on, for instance, *T. domesticum* and also influenced bark beetle composition. The above-mentioned effects of habitat features on beetles help in explaining the importance of management cessation. Furthermore, the effects of forest management on ground beetles are consistent with studies that have highlighted the influence of forestry (e.g. Magura et al. 2003; Niemelä et al. 2007; Paillet et al. 2010) and confirm that not only forest specialist species are favoured by forestry abandonment (Toïgo et al. 2013) but also, in our case, generalist species. Even for bark beetles, several studies have highlighted changes in species composition, richness and abundance between managed and unmanaged forests (Schlyter and Lundgren 1993; Väisänen et al. 1993; Martikainen et al. 1996). Furthermore, saproxylic beetles are generally negatively affected by silvicultural activities (Grove 2002; Paillet et al. 2010). Nevertheless, our study lacks replication due to the low number of relatively large areas of such forests in the Alps, and results presented here should be interpreted with caution. For instance, we recognise that the effects observed here may also be the result of factors other than management, such as historical species distribution and landscape legacy (Sitzia and Trentanovi 2011). Yet, silver fir forests as those sampled here are of special value in terms of biodiversity in Europe (Ellenberg 1988).

6.5 Conclusions

We have shown that ground and saproxylic beetles responded differently to a set of habitat features as a possible result of the abandonment of forest management. For carabid beetles, soil moisture and understorey vegetation cover appear to be of particular importance to their abundance in these forests, while for saproxylic beetles, patterns were more complex with some deadwood features as well as the volumes of standing trees, in particular *A. alba*, playing a role. Our results, even though limited to the geographic area of the Alps, provide preliminary evidence for setting aside forest areas for maintaining and restoring biodiversity in forested landscapes that have been subjected to century-long human alterations. Seizing forest management has a direct effect on those habitat features important to beetle communities and to species linked to old-growth forest characteristics. These habitat features and, therefore, these forests may become readily, or within a relatively short time span, a refuge for many different taxa. In landscapes where abandonment of management and rewilding of forests are occurring due to socio-economic reasons, as is the case for many marginal areas in Europe (Piussi and Farrell 2000; Scarascia-Mugnozza et al. 2000), setting aside seems not to be essential as natural processes are already occurring with consequent changes in habitat features, probably resulting in similar responses in invertebrate communities to those detected here. Moreover, this practice should not be associated with the abandonment of open habitats, which are equally important to maintain biodiversity in mountainous regions (Sitzia et al. 2010).

Acknowledgments

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6.6 Supplementary material

Table S1: References to identification keys used to identify beetle species.

Taxon	References
Ground beetles	Casale A, Sturani M, Vigna Taglianti A (1982) Fauna d'italia. Coleoptera: Carabidae I - Introduzione, Paussinae, Carabinae. Calderini, Bologna.
	Jeannel R (1941) Faune de France 39. Coléoptères Carabiques. Première partie. Librairie de la Faculte des Sciences, Paris.
	Jeannel R (1942) Faune de France 40. Coléoptères carabiques. Deuxiéme partie. Librairie de la Faculte des Sciences, Paris.
	Jeannel R (1949) Faune de France 51. Coléoptères carabiques, supplément. Librairie de la Faculté des Sciences, Paris.
	Mlynar Z (1977) Revision der Arten und Unterarten der Gattung Molops Bon. (s. str.) (Coleoptera, Carabidae). Folia Entomol Hung 30 (Suppl.): 3–150.
	Pesarini C, Monzini V (2010) Insetti della Fauna Italiana - Coleotteri Carabidi I. Nat – Riv Sci Nat 100: 1–152.
	Pesarini C, Monzini V (2011) Insetti della Fauna Italiana - Coleotteri Carabidi II. Nat – Riv Sci Nat 101: 1–144.
	Schatzmayr A (1926) I Trichotichnus (Asmerinx) italiani. Boll Soc Entomol Ital 58: 34–36.
	Schatzmayr A (1929) I Pterostichus italiani. Mem Soc Entomol Ital 8: 145–339.
	Vigna Taglianti A (2005) Appendice B. Checklist e corotipi delle specie di Carabidi della fauna italiana. I Coleotteri Carabidi per la valutazione ambientale e la conservazione della biodiversità. In Brandmayr P, Zetto T, Pizzolotto R (ed) Manuale operativo, APAT, Rome, pp. 186–225.
Bark beetles	Balachowsky A (1949) Faune de France 50. Coleopteres Scolytides. Librairie de la Faculte des Sciences, Paris.
	Colonnelli E (2003) A revised checklist of Italian Curculionoidea (Coleoptera). Zootaxa 337: 1–142.
	Pfeffer A (1994) Zentral- und Westpalaeartische Borken- und Kernkäfer (Coleoptera, Scolytidae, Platypodidae). Entomol Basiliencia 17: 5–310.
	Schedl KE (1981) Familie: Scolytidae (Borken- und Ambrosiakäfer) (Ipidae). In: Freude H, Harde KW, Lohse GA, Die Käfer Mitteleuropas, Vol. 10. Goecke, Evers, Krefeld, pp. 34–99.
	Wood SL (1982) The Bark and Ambrosia beetles of North and Central America (Coleoptera: Scolytidae), a taxonomic monograph. Great Basin Nat Mem 6: 1–1359.
Longhorn beetles	Müller G (1949-53) I coleotteri della Venezia Giulia. Vol II: Coleoptera phytophaga (Cerambycidae, Chrysomelidae, Bruchidae). La Editoriale Libraria, Trieste.
	Pesarini C, Sabbadini A (1994) Insetti della Fauna Europea - Coleotteri Cerambycidae. Nat – Riv Sci Nat 85: 1–132.
	Sama G, Rapuzzi P (2011) Una nuova Checklist dei Cerambycidae d'Italia (Insecta

Coleoptera Cerambycidae). Quad Stud Not Storia Nat Romagna 32: 121–164.

Sama G (1988) Fauna d'Italia. XXVI. Coleoptera Cerambycidae. Catalogo topografico e sinonimico. Calderini, Bologna.

Sama G (1995) Coleoptera Polyphaga XIV (Cerambycidae). In: Minelli A, Ruffo S, La Posta S, Checklist delle specie della fauna italiana 59. Calderini, Bologna, pp 1–12.

Sama G (2002) Atlas of the Cerambycidae of Europe and the Mediterranean Area. Vol. 1, Northern, Western, Central and Eastern Europe. British Isles and Continental Europe from France (excl. Corsica) to Scandinavia and Urals. Nakladatelstvi Kabourek, Zlín.

Table S2: Ground (Carabidae), longhorn (Cerambycidae) and bark beetle (Curculionidae, Scolytinae) species collected. The total number of individuals collected from the managed Cajada and abandoned Tovanella sites are also given.

Species	Abbreviation	Cajada	Tovanella
Carabidae			
Abax parallelepipedus inferior	Abapar	69	96
Abax pilleri	Abapill	196	424
Calathus micropterus	Calmic	0	1
Carabus creutzeri kircheri	Carcre	9	99
Carabus germari	Carger	30	2
Carabus linnaei	Carlin	123	374
Crisimus placidus besucheti	Cripla	8	8
Cychrus angustatus	Cycang	1	1
Cychrus attenatus	Cycatt	8	7
Leistus nitidus	Leinit	25	0
Molops piceus austriacus	Molpic	21	264
Notiophilus biguttatus	Notbig	75	31
Platyderus rufus transalpinus	Plaruf	8	1
Pterostichus burmeisteri	Ptebur	835	1146
Pterostichus oblongopunctatus	Pteobl	2	0
Pterostichus quadrifoveolatus	Ptequa	1	0

Pterostichus unctulatus	Pteunc	254	205
Synuchus vivalis vivalis	Synviv	2	0
Trichotichnus laevicollis	Trilae	15	0
Cerambycidae			
Alosterna tabacicolor	Alotab	3	4
Callidium aeneum	Calaen	0	1
Clytus lama	Clylam	1	0
Grammoptera ruficornis	Graruf	0	1
Leiopus nebulosus	Leineb	1	3
Obrium brunneum	Obrbru	0	4
Phymatodes testaceus	Phytes	0	1
Pidonia lurida	Pidlur	0	2
Pogonocherus hispidulus	Poghis	0	1
Rhagium bifasciatum	Rhabif	0	6
Rhagium inquisitor	Rhainq	1	0
Rhagium mordax	Rhamor	1	9
Rutpela maculata	Rutmac	0	1
Saphanus piceus	Sappic	0	3
Tetropium castaneum	Tetcast	1	3
Curculionidae (Scolytinae)			
Cryphalus abietis	Cryabi	12	9
Crypturgus cinereus	Crycin	0	1
Dryocoetes autographus	Dryaut	13	32
Ernoporicus fagi	Ernfag	1	6
Hylastes cunicularius	Hylcun	74	248
Hylurgops palliatus	Hylpal	0	1

Hypothenemus eruditus	Hyperu	4	15
Ips typographus	Ipstyp	7	2
Phloeotribus spinulosus	Phlspi	1	6
Pityophthorus pityographus	Pitpit	0	1
Polygraphus grandiclava	Polgra	0	1
Scolytus intricatus	Scoint	0	19
Trypodendron domesticum	Trydom	3	103
Trypodendron lineatum	Trylin	7	19
Xyleborinus saxesenii	Xylsax	8	8
Xyleborus dispar	Xyldis	20	68
Xylosandrus germanus	Xylger	16	810

6.7 References

Abrahamsson M, Lindbladh M (2006) A comparison of saproxylic beetle occurrence between man-made high- and low-stumps of spruce (*Picea abies*). For Ecol Manag 226:230–237. doi:10.1016/j.foreco.2006.01.046

Andrich O (2005) Progetto Speciale Selvicoltura e Piano di Riordino Forestale. Parco Nazionale delle Dolomiti Bellunesi, Feltre

Bates D, Maechler M, Bolker B, Walker S (2014) Package 'lme4', Version 1.1-7. http://cran.r-project.org/web/packages/vegan/vegan.pdf

Bengtsson J, Nilsson SG, Franc A, Menozzi P (2000) Biodiversity, disturbances, ecosystem function and management of European forests. For Ecol Manag 132:39–50. doi:10.1016/S0378-1127(00)00378-9

Bouget C, Nusillard B, Pineau X, Ricou C (2012) Effect of deadwood position on saproxylic beetles in temperate forests and conservation interest of oak snags. Insect Conserv Divers 5:264–278. doi:10.1111/j.1752-4598.2011.00160.x

Bouget C, Larrieu L, Parmain G, Nusillard B (2013) In search of the best local habitat drivers for saproxylic beetle diversity in temperate deciduous forests. Biodivers Conserv 22:2111–2130

Bouget C, Parmain G, Gilg O, Noblecourt T, Nusillard B, Paillet Y, Pernot C, Larrieu L, Gosselin F (2014) Does a set aside conservation strategy help the restoration of old-growth forest attributes and recolonization by saproxylic beetles? Anim Conserv 17:342–353. doi:10.1111/acv.12101

Braun-Blanquet J (1932) Plant sociology. Mc-Graw Hill, New York

Brin A, Bouget C, Valladares L, Brustel H (2013) Are stumps important for conservation of saproxylic beetles in managed forests? – Insights from a comparison of assemblages on logs and stumps in oak-dominated forests and pine plantations. Insect Conserv Divers 6:255–264. doi:10.1111/j.1752-4598.2012.00209.x

Brunet J, Fritz Ö, Richnau G (2010) Biodiversity in European beech forests—a review with recommendations for sustainable forest management. Ecol Bull 53:77–94

Castellani C, Scrinzi G, Tabacchi G, Tosi V (1984) Inventario forestale nazionale italiano (I.F.N.I.), Tavole di cubatura a doppia entrata. Istituto Sperimentale per l'Assestamento Forestale e per l'Alpicoltura, Trento

Cobb TP, Langor DW, Spence JR (2007) Biodiversity and multiple disturbances: boreal forest ground beetle (Coleoptera: Carabidae) responses to wildfire, harvesting, and herbicide. Can J For Res 37:1310–1323. doi:10.1139/X06-310

Desender K, Maelfait J-P, Baert L (1991) Carabid beetles as ecological indicators in dune management (Coleoptera:Carabidae). Elytron 5(Suppl.):239–247

Ellenberg H (1988) Vegetation ecology of Central Europe. Cambridge University Press, Cambridge

Ellenberg H, Weber HE, Düll R, Wirth W, Werner W, Paulißen D (1991) Zeigerwerte von Pflanzen in Mitteleuropa. Scr Geobot 18:1–267

Fuller RJ, Oliver TH, Leather SR (2008) Forest management effects on carabid beetle communities in coniferous and broadleaved forests: implications for conservation. Insect Conserv Divers 1:242–252. doi:10.1111/j.1752-4598.2008.00032.x

Green P, Peterken GF (1997) Variation in the amount of dead wood in the woodlands of the Lower Wye Valley, UK in relation to the intensity of management. For Ecol Manag 98:229–238. doi:10.1016/S0378-1127(97)00106-0

Grove JS (2002) Saproxylic insect ecology and the sustainable management of forests. Annu Rev Ecol Syst 33:1–23

Hedgren PO (2007) Early arriving saproxylic beetles (Coleoptera) and parasitoids (Hymenoptera) in low and high stumps of Norway spruce. For Ecol Manag 241:155–161. doi:10.1016/j.foreco.2007.01.020

Hjältén J, Johansson T, Alinvi O, Danell K, Ball JP, Pettersson R, Gibb H, Hilszczański J (2007) The importance of substrate type, shading and scorching for the attractiveness of dead wood to saproxylic beetles. Basic Appl Ecol 8:364–376. doi:10.1016/j.baae.2006.08.003

Hjältén J, Stenbacka F, Andersson J (2010) Saproxylic beetle assemblages on low-stumps, high-stumps and logs: implications for environmental effects of stump harvest. For Ecol Manag 260:1149–1155. doi:10.1016/j.foreco.2010.07.003

Humphrey JW, Hawes C, Pearce AJ, Ferris-Khan R, Jukes MR (1999) Relationships between insect diversity and habitat complexity in plantation forests. For Ecol Manag 113:11–21. doi:10.1016/S0378-1127(98)00413-7

Jonsell M, Hansson J (2011) Logs and stumps in clearcuts support similar saproxylic beetle diversity: implications for bioenergy harvest. Silva Fenn 45:1053–1064. doi:10.14214/sf.86

Jonsell M, Schroeder M, Weslien J (2005) Saproxylic beetles in high stumps of spruce: fungal flora important for determining the species composition. Scand J For Res 20:54–62. doi:10.1080/02827580510008211

Kacprzyk M, Bednarz B (2014) The occurrence of bark beetles on cut Norway spruce branches left in managed stands relative to the foliage and bark area of the branch. J For Res. doi:10.1007/s10310-014-0449-y

Kotze DJ, Brandmayr P, Casale A, Dauffy-Richard E, Dekoninck W, Koivula MJ, Lovei GL, Mossakowski D, Noordijk J, Paarmann W, Pizzolotto R, Saska P, Schwerk A, Serrano J, Szyszko J, Taboada A, Turin H, Venn S, Vermeulen R, Zetto T (2011) Forty years of carabid beetle research in Europe—from taxonomy, biology, ecology and population studies to bioindication, habitat assessment and conservation. Zookeys 100:55–148. doi:10.3897/zookeys.100.1523

Kotze DJ, O'Hara RB, Lehvävirta S (2012) Dealing with varying detection probability, unequal sample sizes and clumped distributions in count data. PLoS ONE 7:e40923. doi:10.1371/journal.pone.0040923

Kula E, Kajfosz R, Polívka J (2011) Cambioxylophagous fauna developing on logging residues of blue spruce (*Picea pungens* Engelmann). J For Sci 57:24–33

Larrieu L, Cabanettes A, Delarue A (2012) Impact of silviculture on dead wood and on the distribution and frequency of tree microhabitats in montane beech-fir forests of the Pyrenees. Eur J For Res 131:773–786. doi:10.1007/s10342-011-0551-z

Lassauce A, Paillet Y, Jactel H, Bouget C (2011) Deadwood as a surrogate for forest biodiversity: meta-analysis of correlations between deadwood volume and species richness of saproxylic organisms. Ecol Indic 11:1027–1039. doi:10.1016/j.ecolind.2011.02.004

Lavorel S, Grigulis K, McIntyre S, Williams NSG, Garden D, Dorrough J, Berman S, Quétier F, Thébault A, Bonis A (2008) Assessing functional diversity in the field—methodology matters! Funct Ecol 22:134–147. doi:10.1111/j.1365-2435.2007.01339.x

Lemperiere G, Marage D (2010) The influence of forest management and habitat on insect communities associated with dead wood: a case study in forests of the southern French Alps. Insect Conserv Divers 3:236–245. doi:10.1111/j.1752-4598.2010.00094.x

Linsley EG (1959) Ecology of cerambycidae. Annu Rev Entomol 4:99–138. doi:10.1146/annurev.en.04.010159.000531

Magura T, Tothmeresz B, Elek Z (2003) Diversity and composition of carabids during a forestry cycle. Biodivers Conserv 12:73–85. doi:10.1023/A:1021289509500

Marshall PI, Davis G, LeMay WM (2000) Using line intersect sampling for coarse woody debris. Vancouver Forest Region, BC Ministry of Forests, Nanaimo

Martikainen P, Siitonen J, Kaila L, Punttila P (1996) Intensity of forest management and bark beetles in non-epidemic conditions: a comparison between Finnish and Russian Karelia. J Appl Entomol 120:257–264. doi:10.1111/j.1439-0418.1996.tb01603.x

Maser C, Anderson R, Cromack JK, Williams JT, Martin RE (1979) Dead and down woody material. In: Thomas JW (ed) Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Wildlife Management Institute, Washington, pp 78–95

Mayer H (1974) Wälder des Ostalpenraumes. Gustav Fisher Verlag, Stuttgart

Motta R, Berretti R, Lingua E, Piussi P (2006) Coarse woody debris, forest structure and regeneration in the Valbona Forest Reserve, Paneveggio, Italian Alps. For Ecol Manag 235:155–163. doi:10.1016/j.foreco.2006.08.007

Mullen K, O'Halloran J, Breen J, Giller P, Pithon J, Kelly T (2008) Distribution and composition of carabid beetle (Coleoptera, Carabidae) communities across the plantation forest cycle— implications for management. For Ecol Manag 256:624–632. doi:10.1016/j.foreco.2008.05.005

Müller J, Bußler H, Kneib T (2008) Saproxylic beetle assemblages related to silvicultural management intensity and stand structures in a beech forest in Southern Germany. J Insect Conserv 12:107–124. doi:10.1007/s10841-006-9065-2

Müller J, Noss RF, Bussler H, Brandl R (2010) Learning from a ‘‘benign neglect strategy’’ in a national park: response of saproxylic beetles to dead wood accumulation. Biol Conserv 143:2559–2569. doi:10.1016/j.biocon.2010.06.024

Niemelä J, Spence JR, Spence DH (1992) Habitat associations and seasonal activity of ground-beetles (Coleoptera, Carabidae) in central Alberta. Can Entomol 124:521–540. doi:10.4039/Ent124521-3

Niemelä J, Haila Y, Punttila P (1996) The importance of small-scale heterogeneity in boreal forests: variation in diversity in forest-floor invertebrates across the succession gradient. Ecography 19:352–368. doi:10.1111/j.1600-0587.1996.tb01264

Niemelä J, Koivula M, Kotze DJ (2007) The effects of forestry on carabid beetles (Coleoptera: Carabidae) in boreal forests. J Insect Conserv 11:5–18. doi:10.1007/s10841-006-9014-0

O’Hara RB, Kotze DJ (2010) Do not log-transform count data. Methods Ecol Evol 1:118–122. doi:10.1111/j.2041-210X.2010. 00021.x

Ódor P, Standovár T (2001) Richness of bryophyte vegetation in near-natural and managed beech stands: the effects of management-induced differences in dead wood. Ecol Bull 49:219–229

Oksanen J, Blanchet FG, Kindt R, Legendre P, Minchin PR, O’Hara RB, Simpson GL, Solymos P, Stevens MHH, Wagner H (2013) Vegan: community ecology package. R package version 2.0-10. http://CRAN.R-project.org/package=vegan

Paillet Y, Bergès L, Hjältén J, Ódor P, Avon C, Bernhardt-Römermann M, Bijlsma RJ, De Bruyn L, Fuhr M, Grandin U, Kanka R, Lundin L, Luque S, Magura T, Matesanz S, Mészáros I, Sebastià MT, Schmidt W, Standovár T, Tóthmérész B, Uotila A, Valladares F, Vellak K, Virtanen R (2010) Biodiversity differences between managed and unmanaged forests: meta-analysis of species richness in Europe. Conserv Biol 24:101–112. doi:10.1111/j.1523-1739.2009.01399.x

Piussi P, Farrell EP (2000) Interactions between society and forest ecosystems: challenges for the near future. For Ecol Manag 132:21–28. doi:10.1016/S0378-1127(00)00376-5

R Development Core Team (2013) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org. Accessed 16 Dec 2013

Rudinsky JA (1962) Ecology of scolytidae. Annu Rev Entomol 7:327–348. doi:10.1146/annurev.en.07.010162.001551

Samways MJ (2007) Insect conservation: a synthetic management approach. Annu Rev Entomol 52:465–487. doi:10.1146/annurev.ento.52.110405.091317

Scarascia-Mugnozza G, Oswald H, Piussi P, Radoglou K (2000) Forests of the Mediterranean region: gaps in knowledge and research needs. For Ecol Manag 132:97–109. doi:10.1016/S0378-1127(00)00383-2

Schielzeth H (2010) Simple means to improve the interpretability of regression coefficients. Methods Ecol Evol 1:103–113. doi:10.1111/j.2041-210X.2010.00012.x

Schlyter F, Lundgren U (1993) Distribution of a bark beetle and its predator within and outside old-growth forest reserves: no increase of hazard near reserves. Scand J For Res 8:246–256. doi:10.1080/02827589309382774

Schmidt W (2005) Herb layer species as indicators of biodiversity of managed and unmanaged beech forests. For Snow Landsc Res 79:111–125

Siitonen J (2001) Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example. Ecol Bull 49:11–41

Similä M, Kouki J, Martikainen P (2003) Saproxylic beetles in managed and seminatural Scots pine forests: quality of dead wood matters. For Ecol Manag 174:365–381. doi:10.1016/S0378-1127(02)00061-0

Sitzia T, Trentanovi G (2011) Maggengo meadow patches enclosed by forests in the Italian Alps: evidence of landscape legacy on plant diversity. Biodivers Conserv 20:945–961. doi:10.1007/s10531-011-0006-3

Sitzia T, Semenzato P, Trentanovi G (2010) Natural reforestation is changing spatial patterns of rural mountain and hill landscapes: a global overview. For Ecol Manag 259:1354–1362. doi:10.1016/j.foreco.2010.01.048

Sitzia T, Trentanovi G, Dainese M, Gobbo G, Lingua E, Sommacal M (2012) Stand structure and plant species diversity in managed and abandoned silver fir mature woodlands. For Ecol Manag 270:232–238. doi:10.1016/j.foreco.2012.01.032

Sollins P (1982) Input and decay of coarse woody debris in coniferous stands in western Oregon and Washington. Can J For Res 12:18–28

Sroka K, Finch O-D (2006) Ground beetle diversity in ancient woodland remnants in north-western Germany (Coleoptera, Carabidae). J Insect Conserv 10:335–350. doi:10.1007/s10841-006-9008-y

Susmel L (1958) Piano di riordinamento della proprietà silvopastorale Costantini in Val Tovanella (1958–1967). Azienda di Stato per le Foreste Demaniali, Belluno

Taboada A, Kotze DJ, Tárrega R, Salgado JM (2006) Traditional forest management: do carabid beetles respond to human-created vegetation structures in an oak mosaic landscape? For Ecol Manag 237:436–449. doi:10.1016/j.foreco.2006.09.077

Taboada A, Kotze DJ, Tárrega R, Salgado JM (2008) Carabids of differently aged reforested pinewoods and a natural pine forest in a historically modified landscape. Basic Appl Ecol 9:161–171. doi:10.1016/j.baae.2007.01.004

Taboada A, Tárrega R, Calvo L, Marcos E, Marcos JA, Salgado JM (2010) Plant and carabid beetle species diversity in relation to forest type and structural heterogeneity. Eur J For Res 129:31–45. doi:10.1007/s10342-008-0245-3

Thiele HU (1977) Carabid beetles in their environments. Springer, Berlin

Toïgo M, Paillet Y, Noblecourt T, Soldati F, Gosselin F, Dauffy-Richard E (2013) Does forest management abandonment matter more than habitat characteristics for ground beetles? Biol Conserv 157:215–224. doi:10.1016/j.biocon.2012.07.025

Ulyshen MD, Hanula JL (2009) Habitat associations of saproxylic beetles in the southeastern United States: a comparison of forest types, tree species and wood postures. For Ecol Manag 257:653–664. doi:10.1016/j.foreco.2008.09.047

Väisänen R, Biström O, Heliövaara K (1993) Sub-cortical Coleoptera in dead pines and spruces: is primeval species composition maintained in managed forests? Biodivers Conserv 2:95–113. doi:10.1007/BF00056127

Van Wagner CE (1982) Practical aspects of the line intersect method. Petawawa National Forestry Institute, Canadian Forest Service, Chalk River

Varady-Szabo H, Buddle CM (2006) On the relationships between ground-dwelling spider (Araneae) assemblages and dead wood in a northern sugar maple forest. Biodivers Conserv 15:4119–4141. doi:10.1007/s10531-005-3369-5

Vuidot A, Paillet Y, Archaux F, Gosselin F (2011) Influence of tree characteristics and forest management on tree microhabitats. Biol Conserv 144:441–450. doi:10.1016/j.biocon.2010.09.030

Winter S, Möller GC (2008) Microhabitats in lowland beech forests as monitoring tool for nature conservation. For Ecol Manag 255:1251–1261. doi:10.1016/j.foreco.2007.10.029

7. Paper VI: Novel woodland patches in a small historical Mediterranean city: Padova, Northern Italy[6]

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Abstract

Woodland fragments, in small historical cities, are commonly regarded as temporary voids in an urban matrix, yet to be allocated a land-use, under city planning regulations. However, they could display relevant plant diversity, and contribute to urban ecosystem services. This study combined surveys at 100 m^2, and at patch level, with the aim to investigate how patch size, stand and urbanization, affected the structure of plant communities in thirty woodland fragments (0.1–2 ha), spontaneously developing in the small, historical city of Padova (Northern Italy). Trees, shrubs and other perennial species dominated the plant communities in these patches. Alien species were common, in both the understory (freq. = 97 %, mean richness = 4.33) and tree layer (freq. = 90 %, mean richness = 1.50). Species typical of native communities also occurred. Understory communities were associated with ancient forest, nitrophilous, and ruderal species; highlighting an overall heterogeneity. Road and railway density was moderately correlated with total species richness in the understory, whereas, urbanity (i.e. the concentration of built environment excluding road and railway density), and tree density were not. Furthermore, alien tree dominance negatively influenced total and native tree layer species richness and, moderately positively, native understory species richness. These results highlight that spontaneous novel woodland patches, even if they are minor fragments in small historical cities, maintain diverse green infrastructures that may supply an array of urban ecosystem services, when adequately recognised by city plans.

Keywords: Biodiversity, Spontaneous afforestation, Urban Planning, Urban forest, Vegetation

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7.1 Introduction

Understanding of urban woodlands is increasing (Croci et al. 2008; Vallet et al. 2010; Trentanovi et al. 2013), this follows significant developments in the field of urban ecology, having understood the services that can be provided by these ecosystems at local, regional, national and global levels (Breuste et al. 2013). Urban woodlands are a product of the uniqueness of the urban environment; there are complex interactions of abiotic and biotic processes (Vallet et al. 2010; Werner 2011) creating ubiquitous ecological conditions (Rebele 1994). Woods are often fragmented in urban areas (Werner 2011); forming patches within the urban matrix, they are one of the most common natural habitats in European towns (Croci et al. 2008). There are different types of canopy, vegetation and plant establishment processes, on abandoned or neglected land and sites disturbed by humans (Kowarik 2005; Mathey and Rink 2010).

In urban landscapes, vegetation is influenced by features of urbanisation, such as, the neighbouring built-up area, surrounding household density (e.g., Kühn and Klotz 2006; McDonnell and Hahs 2008), road and railway density (e.g., von der Lippe and Kowarik 2008; Penone et al. 2012). In turn, these factors are intrinsically shaped by city planning (Sukopp et al. 1995), they mainly regard the development of residential, and industrial areas, in greenfield and brownfield sites, and the related construction of roads. However, in the case of urban forests, other stand-level factors may play an important role in determining the vegetation communities (e.g., Gilliam 2007; Barbier et al. 2008), together with the size of the patch (e.g., Hobbs 1988; Iida and Nakashizuka 1995).

The majority of historical cities are from the Mediterranean basin and Asia; areas subjected to human influence for millennia. Therefore, results from outside these regions may contrast with those observed within. Authors have outlined that hot spots of species richness are found in European cities (e.g., Zerbe et al. 2003; Kowarik 2011), there is a positive relationship between settlements and/or buildings age and plant species composition (Celesti-Grapow et al. 2006).

Spontaneous vegetation can form a part of urban woodlands (Kowarik 2005; Celesti-Grapow et al. 2006) and suburban areas (Barbati et al. 2013); the ecosystem services that can potentially be provided by spontaneous vegetation are considerable, such as habitat provision, wildlife dispersal, climate regulation and carbon sequestration (Robinson and Lundholm 2012; Barbati et al. 2013). The influx of alien species into European cities has been widely documented (e.g., Pyšek 1998; Wittig 2004), an overstory dominated by alien tree species, may have an influence on understory communities (e.g., Richardson et al. 1989; Sitzia et al. 2012). Species which are favoured by human disturbance, and fragmentation, tend to prevail in spontaneous woodland patches (Kowarik 2005; Del Tredici 2010; Kowarik et al. 2013; Trentanovi et al. 2013). Newly established plant assemblages, formed by alien species can be addressed as 'novel ecosystems' (Hobbs et al. 2006; Kowarik 2011).

Current research in Europe on urban woodlands focuses on large cities - for example; Rome (Celesti-Grapow et al. 2006) and Berlin (Trentanovi et al. 2013), or on mixed habitats (e.g., Celesti-Grapow and Blasi 1998; Chocholoušková and Pyšek 2003), woodlands which range from ‘wild’ to managed (Croci et al. 2008), or relatively old and/or studying the effects of rural to urban gradients (Lehvävirta and Rita 2002; Vallet et al. 2010). Species richness in small patches of spontaneous urban woodlands in small cities is relatively unknown, as well as the existence of these novel patches within the urban planning context, and for the people living in and around them.

The purpose of this paper is to develop knowledge on small novel transient woodland patches in a small historical city in Mediterranean Europe, firstly, to understand the composition of plant species assemblages, and secondly, to understand what factors are shaping plant species richness in these patches. Patches were selected randomly, and their boundaries defined. Their main vegetation characteristics were studied, native and alien species were recorded in the understory and tree layers at the sampling plot level (100 m2), and for woody species at the patch level. Then, we assessed the effect of adjacent land-uses, which are commonly allocated in urban plans (i.e., built-up area and road and railway density), and factors important at the stand level (i.e., tree density and alien tree dominance), on the species richness.

7.2 Methods

7.2.1 Study area

The small sized historical city of Padova (English: Padua), founded around X-IX sec. BC, is a municipality of 92.85 km^2 (210,000 residents) with an ancient town centre of 4.54 km2 (Comune di Padova 2012); it is located in the Veneto region, Northeast of Italy (45°23′N, 11° 52′ E). The climate is sub-Mediterranean; the annual precipitation is 846 mm and the mean annual temperature is 12.9°C. Land use is mainly built up urban residential settlements (~55.5 %) with areas mixed with agricultural uses (~41.5 %); most of the territory is modelled artificially. Woody vegetation is very limited and mostly confined to the margins of rivers. According to ARPAV (2013), spontaneous wooded areas in Padova are almost non-existent. Road and railway networks extend for 982 and 31.5 km within the territory, respectively (Comune di Padova 2012).

7.2.2 Data Collection

Sampling was performed from the start of June, to the first weekend of August 2013, with most of the work being performed in July. Within the municipality boundary, we searched woodland patches≥1000 m^2 to avoid those that would have been dominated by edge effects (Matlack 1994). We excluded patches that showed strong active management, or no spontaneous vegetation, or no trees ≥3 cm diameter breast height (DBH), and height ≥5 m. A sample of thirty of these patches were randomly selected (Fig. 1). From observing signs of spontaneous vegetation in historical images, the age of these patches ranged between 10 and 30 years. Only sporadic cuts were observed in certain patches and their boundaries were often limited by human elements or activities, like roads, buildings and cultivation. The interior of these patches was frequently used by people, where shelters and tracks were frequently observed (Fig. 2). A working protocol was adopted to collect information on these patches and to easily enable future replications. In all the patches, all woody species, including seedlings, were identified. Within these patches, a 100 m^2 plot was surveyed, to record the plant species of the understory and tree layer vegetation. The cover-abundance scale of Braun-Blanquet and Pavillard (1928) (r= solitary small individual; +=few individuals; 1=<5 %; 2=5–25 %; 3=25–50 %; 4=50–75 %; 5=>75 %) was used to estimate understory species cover. The basal area was calculated from all trees with DBH ≥3 cm.

Fig. 1 Study area location and distribution of the woodland patches surveyed in the city of Padova

7.2.3 Data analysis

Cover-abundance values of recorded species were converted into Tüxen and Ellenberg (1937) percentage values (r=0.02; +=0.1; 1=2.5; 2=15; 3=37.5; 4=62.5; 5=87.5).

First we identified life-form; life-span and chorological type, using Pignatti (1982) and other databases, for each species recorded in the understory, and then the overall proportion of these life traits was calculated. Secondly, we ascertained that there was not any spatial autocorrelation present in the species richness values running the command Moran.I of the package “ape” in R (Paradis et al. 2004), using the inverse Euclidean distance matrix among patches’ centroids. Then, to investigate drivers of species richness, richness values were calculated at patch, and plot level, for total, native and alien species. In our study, alien species were identified according to Celesti-Grapow et al. (2009) and Masin and Scortegagna (2012) and considering also those species intentionally planted and unlikely to occur spontaneously.

Fig. 2 Pictures showing: the interior of a patch (a), an example of use by people (b), the spontaneous expansion by the alien black locust in the urban-agricultural fringe (c), and along a central urban road (d)

The Bray-Curtis similarity distance matrix, from the understory species cover values at each site, were subjected to agglomerative hierarchical clustering analyses by using Ward's clustering method. We then identified three groups of sites that adequately represented the observed flora. To investigate which plant species defined these groups, we performed Indicator Species Analysis (ISA) (Dufrêne and Legendre 1997). "Multipatt" function of the "indicspecies" package in R software (De Caceres et al. 2010) was used as the indicator value method (IndVal) and the statistical significance was tested with the Monte Carlo test, based on 9999 randomisations. This method combines data on both abundance and frequency. The resulting association index, IndVal.g, is the square root of the IndVal index in Dufrêne and Legendre (1997), it is at maximum (=1) when all occurrences of a species are found in a single group of sites, and when the species occurs in all sites of that group. To understand what factors affect the response of understory species richness, two groups of predictor variables, land use (urbanity, road and rail density) and stand (mean DBH, basal area and alien dominance) characteristics, were used. To calculate the land use variables, Quantum GIS (Quantum GIS Development Team 2012) was used. A 500m buffer was established for each patch, this buffer size has been shown to yield the best predictor set, compared to 100m and 200m buffers (e.g., Knapp et al. 2009; Westermann et al. 2011). Urbanity, as described by Trentanovi et al. (2013), was derived by subtracting rail and road density from urban land-use, based on Corine Land Cover classification (Bossard et al. 2000), calculating its proportion in the buffer. For road and rail density, the total length within the buffer was divided by the total buffer area. Basal area was used because it is a common parameter of tree density and canopy cover, particularly in reference to overstory-understory relationships (Mitchell and Popovich 1997), it is understood that, with increasing basal area, there is a decrease in light transmittance to the forest floor (Korhonen et al. 2006). The alien tree dominance was computed as the proportion of alien species basal area. The nature and strength of covariation between richness values and urbanity, road and rail density, basal area and alien tree dominance was tested through the Pearson product-moment correlation coefficient, after removal of outliers, if any were identified with the diagnostics method performed by the function “influence.measures” in R software (Belsley et al. 1980) or log-transformation of variables with skewed distributions. Where variables exhibited strong departures from normal distribution, we used Spearman rank correlation test. We also investigated the influence of patch size on the total richness of woody species in the patch. Linear regression analysis was used to find the best-fit line relating patch size to woody species richness. R software (Version 3.0.1) (R Development Core Team 2013) was used for all statistical analyses.

7.3 Results

The thirty surveyed patches had a mean size of 0.645 (SD: 0.50; min: 0.12; max: 2.32) ha, a mean tree basal area of 42 (SD: 30.1; min: 6; max: 157) m^2/ha. Yet the highest basal area values were due to the presence of sizeable, old trees, pre-existent to the land use abandonment.

A total of 106 species were identified, of which 38 were alien (Suppl. Material). Table 1 shows the most frequent species recorded at the plot level (total: 92, aliens: 34). Species richness in the understory and tree layer (Table 2), and for woody species at the patch level (Table 3) underlined the importance of the alien component.

Table1 First three most frequent native and alien plant species in thirty 100m^2 plots each representative of a woodland patch of Padova (Northern Italy) urban area

	Alien species	Freq. (%)	Native species	Freq. (%)
Understory	*Robinia pseudoacacia*	60	*Cornus sanguinea*	80
	Laurus nobilis	50	*Rubus fruticosus* agg.	73
	Ligustrum japonicum	47	*Hedera helix*	73
	Acer negundo	47	*Acer campestre*	60
Tree layer	*Robinia pseudoacacia*	67	*Ulmus minor*	40
	Acer negundo	17	*Sambucus nigra*	33
	Prunus cerasifera	13	*Salix alba*	23
			Cornus sanguinea	23

Table 2 Plant species richness (total, native and alien) observed in thirty 100m^2 plots (understory and tree layer) each representative of a woodland patch of Padova (Northern Italy) urban area. Mean % is the mean proportion calculated from each site

Vegetation layers		Total	Native	Alien
Understory	**Mean**	13.70	9.67	4.03
	SD	4.53	3.14	2.06
	Mean %	100	71.74	28.26
	Total	85	56	29
Tree layer	**Mean**	3.57	2.10	1.47
	SD	1.63	1.35	0.82
	Mean %	100	52.90	47.10
	Total	30	17	13

Phanerophytes and perennial species were dominant in the understory. The chorological spectrum highlighted the relevancy of species with Asiatic, additionally to those with European, origin (Fig. 3).

7.3.1 Understory assemblage

The IndVal analysis, conducted for the three groups derived from the cluster analysis, produced a total of eight indicator species (Table 4). Group A was associated with *Hedera* *helix* and *Bryonia dioica*, species common in most types of woodlands and sheltered sites, with preference for heavy, fertile soils (Harding and Hilton 1992; Metcalfe 2005). When occurring in woodland, the *Hedera helix* is frequently dominant in the field layer (see review by Metcalfe 2005), which was also verified here.

Ulmus minor subsp. *minor* was significantly associated with group B, it is a fast growing tree species able to colonise abandoned land, a species which is now re-establishing after gaining resistance to Dutch elm disease (Solla et al. 2005; Sitzia et al. 2012).

Group C was characterised by two ruderal and nitrophilous species: *Rubus fruticosus* agg. and *Parietaria officinalis*, also present was *Brachypodium sylvaticum*; a species regarded by some authors as an ancient forest species (Hermy et al. 1999).

Groups A and C shared a common indicator species, *Sambucus nigra*, a species typical of disturbed, highly eutrophic soils, subjected to disturbance, either naturally, on floodplain terraces and woodland margins, or anthropogenically, in derelict gardens, farmyards and post-industrial wasteland (Atkinson and Atkinson 2002).

7.3.2 Richness and correlated factors

Surprisingly, no relation between species richness, and both urbanity and basal area was found. Road and railway density moderately correlated to understory total ($r=0.41$, $p=0.023$) species richness, but not native or alien species richness. Understory native species richness seemed also to be moderately positively correlated with alien dominance ($r=0.37$, $p=0.042$). As expected, alien tree dominance negatively correlated to species richness, at a limited extent, tree layer total ($\rho=-0.39$, $p=0.035$) and, strongly, native ($\rho=-0.71$, $p<0.001$) species richness.

The diversity of woody species on patches was related to the size of the patch, and the relationship, after removal of two outliers, conformed to: woody species richness=7.179 (patch size in hectares)+13.188 (R^2 adj =0.43, $F=21.515$, $p<0.001$).

Table 3 Woody species richness (total, native and alien) present over the entire area of thirty woodland patches of Padova (Northern Italy) urban area. Mean % is the mean proportion calculated from each site

	Total	Native	Alien
Mean	17.73	10.97	6.77
SD	5.85	3.64	2.76
Mean %	100	61.86	38.14
Total	66	35	31

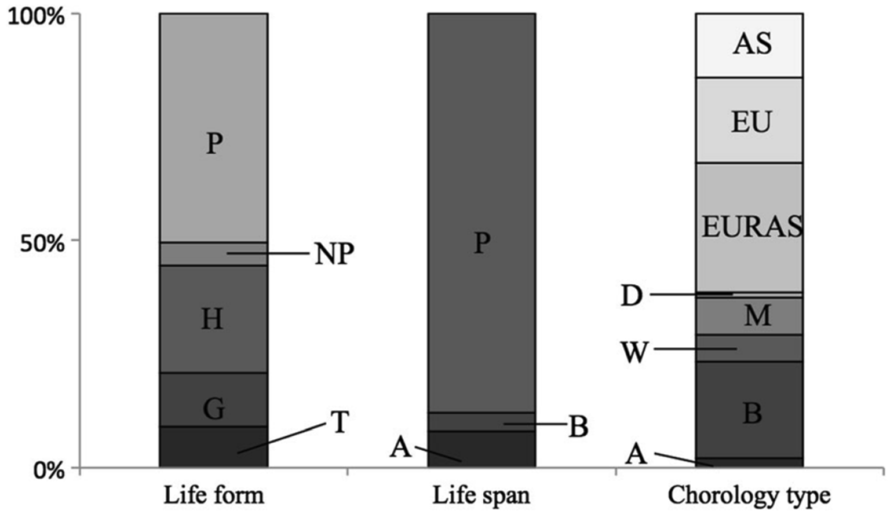

Fig 3 Life traits for the recorded understory species in thirty woodland patches of Padova (Northern Italy). The life form (phanerophytes: P; geophytes: G; hemicryptophytes: H; nano-phanerophytes: NP; therophytes: T), lifespan (annual: A; biannual: B; perennial: P) and chorological type (*AS* asiatic, *EU* European/Caucasic, *EURAS* eurasiatic, *D* dubious origin, *M* Mediterranean, *W* widely-distributed, *B* circumboreal, *A* Atlantic) are reported

7.4 Discussion

Prior work has acknowledged the occurrence, and relevance, of the spontaneous development of vegetation communities in urban areas (Millard 2000, 2004). Nevertheless, studies of biodiversity in the urban environment do not normally use forests as the main foci, or they focus mainly on forest remnants (e.g., Guntenspergen and Levenson 1997; Godefroid and Koedam 2003; Doody et al. 2010). The importance of secondary woodlands spontaneously growing on abandoned lands in urban areas has been recognised (see Kowarik 2005), but studies in southern Europe are lacking and, to date, research has neglected spontaneous forests within small historical urban cities. In this study we have partially filled the research gap, by analysing vegetation assemblages within relatively young, spontaneously developing woodland patches, on neglected land, within a small historical city. We found that, in particular in the tree layer, alien species have an important role, and that the understory shows a relatively heterogeneous composition and richness.

A high proportion, and dominance of alien species, is generally observed in urban plant assemblages (Pyšek 1998; Chocholoušková and Pyšek 2003) and in particular also in wild urban woods (Kowarik 2005). In our context, their high number and relevancy may also be explained by a number of factors: the small patch area, because edge proximity favours alien species both in the understory, and overstory (LaPaix et al. 2012); by the long historical influence of the city, and by the moderate levels of urbanization (McKinney 2008).

As suggested by other studies (e.g., Prach and Pyšek 2001; Celesti-Grapow and Blasi 2003), ours highlights that woody species are important invaders of abandoned urban land. Our study emphasises the dominance of perennial species in the understory. Analysis of the understory composition highlighted the presence of groups of patches characterised by ruderal, and forest-related species. These findings, in accordance with

Kowarik (2011), underline the presence of transient novel forest assemblages, in the studied urban context, a product of inadvertent action by humans, which feature a pronounced alien component. Some patches were also dominated by native trees known to be colonising species, or species typical of the Po plain forests, such as *Ulmus minor*, *Quercus robur*, *Carpinus betulus* and *Salix alba*. This heterogeneity of the composition, both in the tree layer and understory, indicates the high value of spontaneous woodland patches for biodiversity in the urban matrix.

Table 4 Indicator species of three relevé groups identified by cluster analysis of the understory in thirty 100m^2 plots in woodland patches of Padova (Northern Italy) urban area, and related indicator (IndVal.g) and p-values (A: 11, B: 8 and C: 11 plots)

Relevé group	Species	IndVal.g	p.value
A	*Hedera helix*	0.926	0.0001
	Bryonia dioica	0.525	0.0271
B	*Ulmus minor* subsp. *minor*	0.68	<0.0001
C	*Rubus fruticosus* agg.	0.542	0.0027
	Parietaria officinalis	0.481	0.0109
	Brachypodium sylvaticum	0.447	0.0294
C+A	*Sambucus nigra*	0.463	0.0234

As highlighted for parks and woodlots in general (Alvey 2006), even in small spontaneous patches, woody species richness tends to increase with patch size. The limited relationship between explanatory variables and the understory and tree layer plant richness strengthens the discourse that the city is an integrated ecosystem (Rebele 1994); where it is not possible to define one variable in particular that is affecting species richness. There is a degree of chaos theory involved, promoting heterogeneity. Richness is a result of complex interactions of abiotic and biotic processes at different scales (Werner 2011).

The fact that we did not consider forest edge and forest interior as two distinct habitats hinders any quantitative assessment of edge effect which are probable in small woodlands (Gonzalez et al. 2010). However, in addition to road and railway density, and alien dominance, as highlighted by our results, it is possible that other factors may play a role, even obscuring, or filtering, those that we analysed, such as, habitat history and configuration (De Sanctis et al. 2010), which, in turn, may influence edge effects and seed source availability from neighbouring roadside and river vegetation (von der Lippe et al. 2008; Säumel and Kowarik 2010), or residential gardens. Edge effects are related to the adjacent urban matrix and forest trails and paths (LaPaix et al. 2012), a higher number of alien species have been found in forest edges adjacent to urban areas compared to agricultural areas (Gonzalez-Moreno et al. 2013).

Edges and their aspect effects light radiation to the forest floor, influencing species composition and richness. North-facing forest margins may not show edge effects on plant composition, or, its spatial penetration within forests may be reduced, while edge effects may be pronounced in south-facing aspect margins (Hamberg et al. 2010). Recreational pressure, presence of paths, related trampling, concentration of nutrients and pollutants, and other kinds of microhabitat features might also explain a portion of the observed species richness variability (Malmivaara-Lämäs et al. 2008; Hamberg et al. 2010). Furthermore, as colonisation is an on-going process, transient communities, and the factors shaping these communities, may be better explained by a study with a longer temporal scale (Lososová et al. 2012).

This study reveals that in small historical cities, small woodland patches that spontaneously develop within an urban landscape can play an important role for biodiversity, by forming both new communities of species, which did not exist in the past, and native habitats that had previously disappeared. They may have the opportunity to convey a set of services; the spatial concentration of people in cities increases the demand for ecosystem services (McDonald 2015). Urban trees and woodlands have a wide range of benefits and uses (e.g., Konijnendijk 2008; Jim and Chen 2009; Escobedo et al. 2011), such as carbon sequestration and the positive effects on humans' wellbeing. Small spontaneous forest patches may act as an added value in respect to the existing and recognised green infrastructures of small urban parks (Nordh and Østby 2013) and roadside wild vegetation (Weber et al. 2014), enhancing ecosystem services in small historical cities, in particular those related to recreation. Given the small size of woodlands, it is feasible to plan recreational trails at fine-scale resolutions, and carefully assess their environmental impact (Sitzia et al. 2014). The acknowledgement of spontaneous woodland patches is important for the planning and development in small sized cities. The opportunities provided by spontaneous, "unofficial' vegetation; described by Mabey (1973) or "unintentional"; as by Kühn (2006), vegetation for urban landscape design is now being understood, with novel perspectives and innovative approaches being proposed and applied (Millard 2000; Prach and Pyšek 2001; Kowarik and Langer 2005; Kühn 2006; Ignatieva et al. 2010). A key problem is the anomaly of these patches in town planning; they can be subjected to land use change before their value is fully comprehended. Temporary measures, suitable for these habitats, could be implemented in greenfield and brownfield sites, potentially on a medium term basis, until the foreseen building development is realised (see Kattwinkel et al. 2011).

Our study underlines the great potential for urban planning, and the importance of plant communities in small historical cities, further research is needed on vegetation composition and dynamics. Specifically, future studies will help to increase understanding of how these novel transient woodlands will develop, positing what potential vegetation communities can be hosted by these unmanaged woodlands. Furthermore, as forest succession on abandoned urban land is common and brownfields rehabilitation is an option in many small cities in Europe (Acosta et al. 2005; Lafortezza et al. 2008), in North America (Greenberg et al. 2001), and also likely in other regions of the world where forests are spontaneously expanding (see review by Sitzia et al. 2010), it is important to trigger the awareness of planners and designers of the opportunities that these areas can give. A call, for new planning approaches, for transient novel woodlands in small historical cities, is required.

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7.5 Supplementary material

List of plant species and their frequency in thirty woodland patches of Padova (Northern Italy) urban area, in 100m^2 plots (understory and tree layer) and at the patch level (woody species). Alien species were identified according to Celesti-Grapow et al. (2009) and Masin and Scortegagna (2012) and considering also those species intentionally planted and unlikely to occur spontaneously.

Species name	Native (N) or alien (A)	Frequency (100m^2 plot)		Frequency (patch)
		Understory	Tree layer	
Acacia dealbata Link	A		0.03	0.03
Acer campestre L.	N	0.60	0.20	0.73
Acer negundo L.	A	0.47	0.17	0.57
Acer platanoides L.	N			0.03
Acer pseudoplatanus L.	N	0.07	0.03	0.27
Acer saccharinum L.	A	0.03		0.10
Aegopodium podagraria L.	N	0.13		
Ailanthus altissima (Mill.) Swingle	A	0.03		0.07
Alnus glutinosa (L.) Gaertn.	N			0.03
Arum maculatum L.	N	0.03		
Avena fatua L.	N	0.07		
Aristolochia clematitis L.	N	0.07		
Phyllostachys edulis (Carrière) J. Houz.	A			0.03
Brachypodium sylvaticum (Huds.) P. Beauv.	N	0.20		
Bryonia dioica Jacq.	N	0.13		
Buxus sempervirens L.	A	0.03		0.07
Carex acutiformis Ehrh.	N	0.03		
Carpinus betulus L.	N	0.03	0.03	0.03
Carex pendula Huds.	N	0.07		
Celtis australis L.	N	0.17	0.03	0.33
Celtis occidentalis L.	A	0.03	0.03	0.03
Cercis siliquastrum L.	N			0.03

Cirsium arvense (L.) Scop.	N	0.03		
Cirsium vulgare (Savi) Ten.	N	0.03		
Chaerophyllum temulum L.	N	0.07		
Chelidonium majus L.	N	0.13		
Clematis vitalba L.	N	0.07		0.17
Convolvulus arvensis L.	N	0.13		
Corylus avellana L.	N	0.03	0.03	0.07
Cornus sanguinea L.	N	0.80	0.23	0.87
Crataegus monogyna Jacq.	N	0.43	0.03	0.63
Cynodon dactylon (L.) Pers.	N	0.03		
Equisetum telmateia Ehrh.	N	0.07		
Erigeron annuus (L.) Pers.	A	0.07		
Euonymus europaeus L.	N	0.10		0.10
Ficus carica L.	N	0.07	0.03	0.23
Fraxinus excelsior L.	N	0.03		0.03
Galium aparine L.	N	0.13		
Glechoma hederacea L.	N	0.10		
Geum urbanum L.	N	0.07		
Hedera helix L.	N	0.73		0.90
Hordeum murinum L.	N	0.07		
Humulus lupulus L.	N	0.37		
Hypericum perforatum L.	N	0.07		
Ilex aquifolium L.	A	0.03		0.03
Juglans nigra L.	A	0.13		0.23
Juglans regia L.	A	0.20	0.03	0.33
Laurus nobilis L.	A	0.50		0.63
Ligustrum lucidum W.T. Aiton	A	0.07		0.20
Ligustrum ovalifolium Hassk.	A	0.03	0.03	0.10
Ligustrum sinense Lour.	A	0.47	0.03	0.47
Lonicera japonica Thunb.	A	0.33		0.50
Lotus corniculatus L.	N	0.03		

Mahonia aquifolium (Pursh) Nutt.	A	0.03		0.07
Malva sylvestris L.	N	0.03		
Morus alba L.	A	0.10	0.07	0.20
Morus nigra L.	A	0.10	0.10	0.30
Oxalis corniculata L.	N	0.10		
Paliurus spina-christi Mill.	N	0.03		0.03
Parietaria officinalis L.	N	0.33		
Parthenocissus quinquefolia (L.) Planch.	A	0.10		0.37
Picea abies (L.) Karst.	A	0.03	0.03	0.13
Phytolacca americana L.	A	0.03		
Platanus acerifolia (Aiton) Willd.	A		0.10	0.40
Poa trivialis L. subsp. *sylvicola* (Guss.) H. Lindb.	N	0.03		
Poa trivialis L. subsp. *trivialis*	N	0.13		
Populus alba L.	N	0.10	0.07	0.33
Populus X canescens (Aiton) Sm.	N			0.13
Populus X canadensis Moench	A		0.03	0.13
Populus nigra L.	N		0.17	0.53
Populus tremula L.	N		0.03	0.10
Duchesnea indica (Andrews) Focke	A	0.10		
Potentilla reptans L.	N	0.03		
Prunus armeniaca L.	A	0.03		0.13
Prunus avium (L.) L.	N	0.30	0.17	0.33
Prunus cerasifera Ehrh.	A	0.37	0.13	0.53
Prunus laurocerasus L.	A	0.07		0.07
Prunus mahaleb L.	N	0.03		0.03
Prunus spinosa L.	N			0.13
Pyracantha coccinea M. Roem.	N	0.03		0.03
Quercus cerris L.	A		0.03	0.03
Quercus robur L.	N	0.40	0.03	0.43
Ranunculus flammula var. *reptans* (L.) E. Mey.	N	0.03		
Ribes rubrum L.	A			0.03

Robinia pseudoacacia L.	A	0.60	0.67	0.77
Rosa canina L.	N			0.13
Rubus fruticosus L. agg.	N	0.73		0.97
Rumex obtusifolius L.	N	0.07		
Salix alba L.	N		0.23	0.63
Salix babylonica L.	A			0.03
Sambucus nigra L.	N	0.57	0.33	0.87
Sicyos angulatus L.	A	0.07		
Silene latifolia Poir.	N	0.10		
Sonchus oleraceus L.	N	0.20		
Symphoricarpos albus (L.) S.F. Blake	A			0.07
Tilia platyphyllos Scop.	N			0.03
Trachycarpus fortunei (Hook.) H. Wendl.	A	0.10		0.13
Ulmus minor Mill. subsp. *minor*	N	0.50	0.40	0.63
Urtica dioica L.	N	0.27		
Veronica persica Poir.	A	0.03		
Viburnum lantana L.	N			0.03
Viola reichenbachiana Jord. ex Boreau	N	0.03		
Vitis vinifera L.	A	0.20		0.40
Wisteria sinensis (Sims) DC.	A	0.03		0.10

7.6 References

Acosta A, Carranza LM, Giancola M (2005) Landscape change and ecosystem classification in a Municipal district of small city (Isernia, Central Italy). Environ Monit Assess 108:323–335

Alvey AA (2006) Promoting and preserving biodiversity in the urban forest. Urban For Urban Green 5:195–201

ARPAV (2013) Pollini e spore fungine allergenici nel Veneto—Rapporto anno 2012. http://www.arpa.veneto.it/temi-ambientali/pollini/file-e-allegati/rapporto-2012/RAPPORTO_Pollini_2012.pdf. Accessed 30 January 2014

Atkinson MD, Atkinson E (2002) *Sambucus nigra* L. J Ecol 90:895–923

Barbati A, Corona P, Salvati L, Gasparella L (2013) Natural forest expansion into suburban countryside: gained ground for a green infrastructure? Urban For Urban Green 12:36–43

Barbier S, Gosselin F, Balandier P (2008) Influence of tree species on understory vegetation diversity and mechanisms involved – a critical review for temperate and boreal forests. For Ecol Manag 254:1–15

Belsley DA, Kuh E, Welsch RE (1980) Regression diagnostics. Wiley, New York

Bossard M, Feranec J, Otahel J (2000) CORINE land cover technical guide—Addendum 2000. European Environmental Agency, Technical Report No. 40, Copenhagen, Denmark. http://www.eea.eu.int. Accessed 12 Dicember 2013

Braun-Blanquet J, Pavillard J (1928) Vocabulaire de sociologie végétale. Imprimerie Roumégous & Déhan, Montpellier

Breuste J, Haase D, Elmqvist T (2013) Urban landscapes and ecosystem services. In: Wratten S, Sandhu H, Cullen R, Costanza R (eds) Ecosystem services in agricultural and urban landscapes. Wiley, Chichester, pp 83–104

Celesti-Grapow L, Blasi C (1998) A comparison of the urban flora of different phytoclimatic regions in Italy. Glob Ecol Biogeogr 7:367–378

Celesti-Grapow L, Blasi C (2003) Archaeological sites as areas for biodiversity conservation in cities: the spontaneous vascular flora of the Caracalla Baths in Rome. Webbia 58:77–102

Celesti-Grapow L, Pyšek P, Jarošík V, Blasi C (2006) Determinants of native and alien species richness in the urban flora of Rome. Divers Distrib 12:490–501

Celesti-Grapow L, Alessandrini A, Arrigoni PV, Banfi E, Bernardo L, Bovio M, Brundu G, Cagiotti MR, Camarda I, Carli E, Conti F, Fascetti S, Galasso G, Gubellini L, La Valva V, Lucchese F, Marchiori S, Mazzola P, Peccenini S, Poldini L, Pretto F, Prosser F, Siniscalco C, VillaniMC, Viegi L,Wilhalm T, Blasi C (2009) Inventory of the non-native flora of Italy. Plant Biosyst 143:386–430

Chocholoušková Z, Pyšek P (2003) Changes in composition and structure of urban flora over 120 years: a case study of the city of Plzeň. Flora 198:366–376

Croci S, Butet A, Georges A, Aguejdad R, Clergeau P (2008) Small urban woodlands as biodiversity conservation hot-spot: a multi-taxon approach. Landsc Ecol 23:1171–1186

De Caceres M, Legendre P, Moretti M (2010) Improving indicator species analysis by combining groups of sites. Oikos 119:1674–1684

De Sanctis M, Alfò M, Attore F, Francesconi F, Bruno F (2010) Effect of habitat configuration and quality on species richness and distribution in fragmented forest patches near Rome. J Veg Sci 21:55–65

Del Tredici PD (2010) Spontaneous urban vegetation: reflections of change in a globalized world. Nat Cult 5: 299–315

Development Core Team R (2013) R: A language and environment for statistical computing. R foundation for statistical computing, Vienna

Comune di Padova (2012) Annuario statistico comunale 2012. www.padovanet.it. Accessed 12 December 2013

Doody BJ, Sullivan JJ, Meurk CD, Stewart GH, Perkins HC (2010) Urban realities: the contribution of residential gardens to the conservation of urban forest remnants. Biodivers Conserv 19:1385–1400

Dufrêne M, Legendre P (1997) Species assemblages and indicator species: the need for flexible assymetrical approach. Ecol Monog 67:345–366

Escobedo FJ, Kroeger T,Wagner JE (2011) Urban forests and pollution mitigation: analyzing ecosystem services and disservices. Environ Pollut 159:2078–2087

Gilliam FS (2007) The ecological significance of the herbaceous layer in temperate forest ecosystems. Bioscience 57:845–858

Godefroid S, Koedam S (2003) How important are large vs. small forest remnants for the conservation of the woodland flora in an urban context? Glob Ecol Biogeogr 12:287–298

Gonzalez M, Ladet S, Deconchat M, Cabanettes A, Alard D, Balent G (2010) Relative contribution of edge and interior zones to patch size effect on species richness: an example for woody plants. For Ecol Manag 259: 266–274

Gonzalez-Moreno P, Pino J, Gassó N, Vilá M (2013) Landscape context modulates alien plant invasion in Mediterranean forest edges. Biol Inv 15:547–557

Greenberg M, Lowrie K, Mayer H, Miller KT, Solitare L (2001) Brownfield redevelopment as a smart growth option in the United States. Environmentalist 21:129–143

Guntenspergen GR, Levenson JB (1997) Understory plant species composition in remnant stands along an urban-to-rural lan-use gradient. Urban Ecosyst 1:155–169

Hamberg L, Fedrowitz K, Lehvävirta S, Kotze DJ (2010) Vegetation changes at sub-xeric urban forest edges in Finland—the effects of edge aspect and trampling. Urban Ecosyst 13:583–603

Harding DJ, Hilton GM (1992) Functional ecology of woodlands and forests. Springer, Berlin

Hermy M, Honnay O, Firbank L, Grashof-Bokdam C, Lawesson JE (1999) An ecological comparison between ancient and other forest plant species of Europe, and the implications for forest conservation. Biol Conserv 91:9–22

Hobbs ER (1988) Species richness in urban forest patches and implications for urban landscape diversity. Landsc Ecol 1:141–152

Hobbs RJ, Arico S, Aronson J, Baron JS, Bridgewater P, Cramer VA, Epstein PR, Ewel JJ, Klink CA, Lugo AE, Norton D, Ojima D, Richardson DM, Sanderson EW, Valladares F, Vilà M, Zamora R, Zobel M (2006) Novel ecosystems: theoretical and management aspects of the new ecological world order. Glob Ecol Biogeogr 15:1–7

Ignatieva M, Stewart GH, Meurk C (2010) Planning and design of ecological networks in urban areas. Landsc Ecol Eng 7:17–25

Iida S, Nakashizuka T (1995) Forest fragmentation and its effect on species diversity in suburban coppice forests in Japan. For Ecol Manag 73:197–210

Jim CY, Chen WY (2009) Ecosystem services and valuation of urban forests in China. Cities 26:187–194

Kattwinkel M, Bierdermann R, Kleyer M (2011) Temporary conservation for urban biodiversity. Biol Conserv 144:2335–2343

Knapp S, Kühn I, Bakker JP, KleyerM, Klotz S, OzingaWA, Poschlod P, Thompson K, ThuillerW, Römermann C (2009) How species traits and affinity to urban land use control large-scale species frequency. Divers Distrib 15:533–546

Konijnendijk CC (2008) The forest and the city: the cultural landscape of urban woodland. Springer, Berlin

Korhonen L, Korhonen KT, RautiainenM, Stenberg P (2006) Estimation of forest canopy cover: a comparison of field measurement techniques. Silva Fenn 40:577–588

Kowarik I (2005) Wild urban woodlands: towards a conceptual framework. In: Kowarik I, Körner S (eds) Wild urban woodlands. new perspectives for urban forestry. Springer, Berlin, pp 1–32

Kowarik I (2011) Novel urban ecosystems, biodiversity, and conservation. Environ Pollut 159:1974–1983

Kowarik I, Langer A (2005) Natur-park Südgel.nde: linking conservation and recreating in an abandoned rail yard in Berlin. In: Kowarik I, Körner S (eds) Wild urban woodlands. new perspectives for urban forestry. Springer, Berlin, pp 287–299

Kowarik I, von der Lippe M, Cierjacks A (2013) Prevalence of alien versus native species of woody plants in Berlin differs between habitats and at different scales. Preslia 85:113–132

Kühn N (2006) Intentions for the unintentional: spontaneous vegetation as the basis for innovative planting design in urban areas. J Landsc Archit 1:46–53

Kühn I, Klotz S (2006) Urbanization and homogenization—comparing the floras of urban and rural areas in Germany. Biol Conserv 127:292–300

Lafortezza R, Corry RC, Sanesi G, Brown RD (2008) Visual preference and ecological assessments for designed alternative brownfield rehabilitations. J Environ Manag 89:257–269

LaPaix R, Harper K, Freedman B (2012) Patterns of exotic plants in relation to anthropogenic edges within urban forest remnants. Appl Veg Sci 15:525–535

Lehvävirta S, Rita H (2002) Natural regeneration of trees in urban woodlands. J Veg Sci 13:57–66

Lososová Z, Chytrý M, Tichý L, Danihelka J, Fajmon K, Hájek O, Kintrová K, Láníková D, Otýpková Z, Řehořek V (2012) Biotic homogenization of Central European urban floras depends on residence time of alien species and habitat types. Biol Conserv 145:179–184

Mabey R (1973) The unofficial countryside. William Collins Sons and Co, Glasgow

Malmivaara-Lämäs M, Löfström HL, Vanha-Majamaa I, Niemelä J (2008) Trampling tolerance of understorey vegetation in different hemiboreal urban forest site types in Finland. Urban Ecosyst 11:1–16

Masin Rizzieri R, Scortegagna S (2012) Flora alloctona del Veneto centro-meridionale (Province di Padova, Rovigo, Venezia e Vicenza—Veneto - NE Italia). Nat Vicentina 15:5–54

Mathey J, Rink D (2010) Urban Wastelands—a change for biodiversity in cities? Ecological aspects, social perceptions and acceptance of wilderness by residents. In: Müller N, Werner P, Kelcey JG (eds) Urban biodiversity and design. Wiley-Blackwell, Hoboken, New Jersey, pp 406–424

Matlack GR (1994) Vegetation dynamics of forest edge—trends in space and successional time. J Ecol 82:113–123

McDonald RI (2015) The effectiveness of conservation interventions to overcome the urban-environmental paradox. Ann N Y Acad Sci. doi:10.1111/nyas.12752

McDonnell M, Hahs A (2008) The use of gradient analysis studies in advancing our understanding of the ecology of urbanizing landscapes: current status and future directions. Landsc Ecol 23:1143–1155

McKinney ML (2008) Effects of urbanization on species richness: a review of plants and animals. Urban Ecosyst 11:161–176

Metcalfe DJ (2005) *Hedera helix* L. J Ecol 93:632–648

Millard A (2000) The potential role of natural colonisation as a design tool for urban forestry—a pilot study. Landsc Urban Plan 52:173–180

Millard A (2004) Indigenous and spontaneous vegetation: their relationship to urban development in the city of Leeds, UK. Urban For Urban Green 3:39–47

Mitchell JE, Popovich SJ (1997) Effectiveness of basal area for estimating canopy cover of ponderosa pine. For Ecol Manag 95:45–51

Nordh H, Østby K (2013) Pocket parks for people—a study of park design and use. Urban For Urban Green 12: 12–17

Paradis E, Claude J, Strimmer K (2004) APE: analyses of phylogenetics and evolution in R language. Bioinformation 20:289–290

Penone C, Machon N, Julliard R, Le Viol I (2012) Do railway edges provide functional connectivity for plant communities in an urban context? Biol Conserv 148:126–133

Pignatti S (1982) Flora d'Italia. Edagricole, Bologna

Prach K, Pyšek P (2001) Using spontaneous succession for restoration of human-disturbed habitats: experience from Central Europe. Ecol Eng 17:55–62

Pyšek P (1998) Alien and native species in Central European urban floras: a quantitative comparison. J Biogeogr 25:155–163

Quantum GIS Development Team (2012) Quantum GIS geographic information systen. open source geospatial foundation project. http://qgis.osgeo.org

Rebele F (1994) Urban ecology and special features of urban ecosystems. Glob Ecol Biogeogr Lett 4:173–187

Richardson DM, MacDonald IAW, Forsyth GG (1989) Reduction in plant species richness under stands of alien trees and shrubs in the Fynbos Biome. South Afr For J 149:1–8

Robinson S, Lundholm J (2012) Ecosystem services provided by urban spontaneous vegetation. Urban Ecosyst 15:545–557

Säumel I, Kowarik I (2010) Urban rivers as dispersal corridors for primarily wind-dispersed invasive tree species. Landsc Urban Plan 94:244–249

Sitzia T, Semenzato P, Trentanovi G (2010) Natural reforestation is changing spatial patterns of rural mountain and hill landscapes: a global overview. For Ecol Manag 259:1354–1362

Sitzia T, Campagnaro T, Dainese M, Cierjacks A (2012) Plant species diversity in alien black locust forests: a paired comparison with native stands across a north-Mediterranean range expansion. For Ecol Manag 285: 85–91

Sitzia T, Rizzi A, Cattaneo D, Semenzato P (2014) Designing recreational trails in a forest dune habitat using least-cost path analysis at the resolution of visitor sight distance. Urban For Urban Green 13:861–868

Solla A, Bohnens J, Collin E, Diamandis S, Franke A, Gil L, Burón M, Santini A, Mittempergher L, Pinon J, Broeck AV (2005) Screening European elms for resistance to *Ophiostoma novo-ulmi*. For Sci 51:134–141

Sukopp H, Numata M, Huber A (1995) Urban ecology as the basis of urban planning. SPB Academic Publishing, Amsterdam

Trentanovi G, von der Lippe M, Sitzia T, Ziechmann U, Kowarik I, Cierjacks A (2013) Biotic homogenization at the community scale: disentangling the roles of urbanization and plant invasion. Divers Distrib 19:738–748

Tüxen R, Ellenberg H (1937) Der systematische und ökologische gruppenwert. ein beitrag zur begriffsbildung und methodik der pflanzensoziologie. Mitt Florist-Soziol Arbgem 3:171–184

Vallet J, Daniel H, Beaujouan V, Rozé F, Pavoine S (2010) Using biological traits to assess how urbanization filters plant species of small woodlands. Appl Veg Sci 13:412–424

von der Lippe M, Kowarik I (2008) Do cities export biodiversity? traffic as dispersal vector across urban-rural gradients. Divers Distrib 14:18–25

Weber F, Kowarik I, Säumel I (2014) Awalk on the wild side: Perceptions of roadside vegetation beyond trees. Urban For Urban Green 13:205–212

Werner P (2011) The ecology of urban areas and their functions for species diversity. Landsc Ecol Eng 7:231–240

Westermann JR, vor der Lippe M, Kowarik I (2011) Seed traits, landscape and environmental parameters as predictors of species occurrence in fragmented urban railway habitats. Basic Appl Ecol 12:29–37

Wittig R (2004) The origin and development of the urban flora of Central Europe. Urban Ecosyst 7:323–329

Zerbe S, Maurer U, Schmitz S, Sukopp H (2003) Biodiversity in Berlin and its potential for nature conservation. Landsc Urban Plan 62:139–148

8. Conclusion

This thesis has highlighted that conservation-sound management is fundamental to maintain the variety of habitats, both natural and semi-natural, occurring in Europe. On the one hand novel approaches, such as those presented in the thesis, are required to face the never-ending changes in legal, economic, social and environmental conditions. On the other hand, deep knowledge on the effects of management and planning choices on habitats and species is essential to achieve conservation goals by adapting to biodiversity's intrinsic variability and complexity. These aspects can be summarized with the most common words used within all the six scientific papers (Fig. 5).

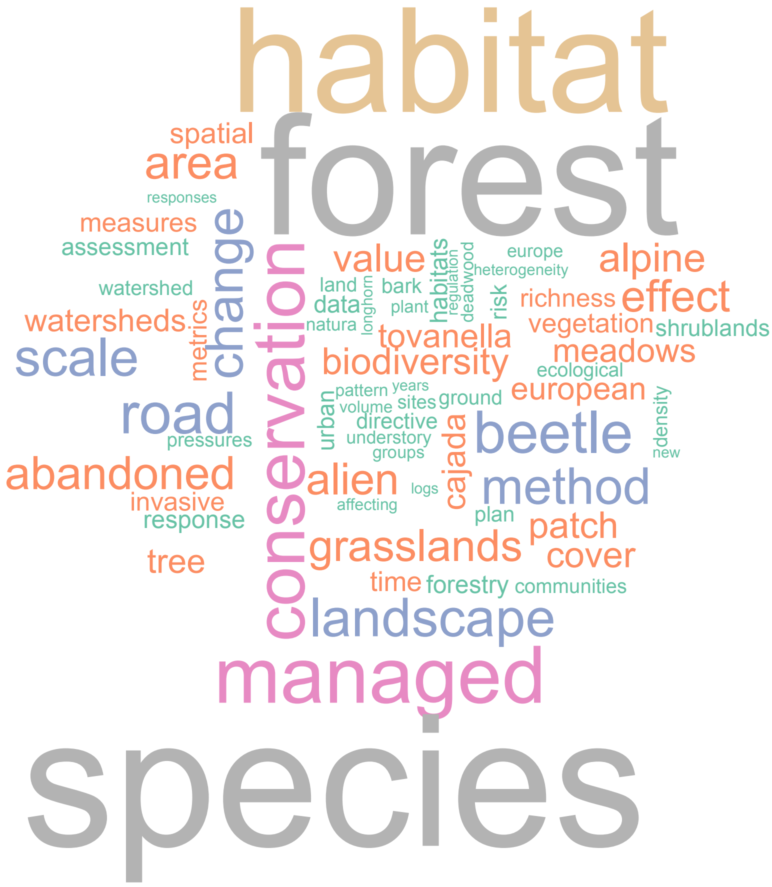

Figure 5: Most frequent terms used in the six papers forming the bulk of this thesis with a text mining analysis[7].

The proposed approach to identify conservation priorities enabled to point out for the Italian alpine and continental biogeographical regions the habitats most in need of management efforts (Paper I). This method by highlighting the main pressures on these habitats gives the possibility of maintaining and improving habitats conservation. Here,

[7] This figure derives from a text mining analysis performed in R statistical software (R Development Core Team, 2015). All the chapters (published and in preparation papers) were pooled together (excluding references, acknowledgements and supplementary materials) and the overall text was transformed to exclude stop words and not important terms using the tm package (Feinerer and Hornik, 2015). The wordcloud function of the wordcloud package (Fellows, 2013) was applied to produce a graphic representation of the most common words.

forest habitats are a key component of biodiversity for which conservation and management is needed. The approach will enable public officers and practitioners to develop and apply appropriate planning and management to achieve biodiversity targets.

Human pressures as highlighted in several chapters of the thesis are posing detrimental effects on species and habitats. However, appropriate methods, such as the one tested in this thesis (Paper II) are needed to understand the real threat posed to biodiversity. The test of this method enabled to indicate that biodiversity conservation could be maintained and achieved even when human development plans are implemented.

Indeed, knowledge and experience in applying management practices can be a tool to face novel biodiversity issues. In this thesis, forest management has been proposed as an approach to tackle invasive alien species spread and impacts (Paper III). However, future studies and the collection of already published research are needed to understand this potential.

A landscape perspective on management effects enables an understanding of trade-offs between different habitats. Multi-scale analysis, as the one presented in this thesis (paper IV), is fundamental to understand cross-scale dynamics in habitat changes and their possible effects on species. However, research efforts are required to shed light on cause-effect relationships for a variety of management practices and taxa. Management cessation but also low-intensity management can favour forest expansion at the expenses of other habitats of conservation value, as was witnessed in this thesis.

When focusing on a specific habitat, different management regimes have a variety of effects on animal communities. For example, forest management abandonment influences differently species and groups of species (paper V) highlighting that a single management approach (including also no management) shapes habitat suitability for a set of species of one taxon and that habitat features should receive a special management focus. However, this paper has highlighted the variety of positive effects linked to forest management abandonment on ground, longhorn and bark beetles.

Management (including its abandonment) can enable the development of novel forests. The research conducted in an urban context (paper VI) shows how even small forests are an opportunity for the recovery of habitats and biodiversity. Furthermore, in this context invasive alien species, bearing in mind both their positive and negative effects, may offer a training ground for innovative management and planning of natural resources.

The multi-scale and multi-taxon approach taken here enabled a holistic overview of conservation needs and possible management solutions. Reported results are sometimes contrasting, but this confirms the complexity of the issue of managing habitats and species to achieve their conservation. Nevertheless, awareness of this complexity, in terms of taxon specific and scale influenced responses, enables an understanding of how to adapt management approaches.

In addition to the research needs highlighted in the single papers, a thorough overview of the thesis makes it possible to outline multiple pathways for future research with respect to:

- Collecting information and data from already published research to enable generalisation of management implications for biodiversity. One important gaps is the lack of reviews on forest habitats management identifying those habitats dependent or partially dependent on management practices (e.g. silviculture systems); for example, it may be possible to apply a similar approach as that in Halada *et al.* (2011), in which they identified habitats linked to agricultural practices.
- Translating the wide knowledge on plants and animals gained through conventional research and historical datasets into “Habitats Directive” lexicon. For example, two recent reviews (Blicharska *et al.*, 2016; Orlikowska *et al.*, 2016) highlighted several research gaps for European protected areas (i.e. Natura 2000 sites) both in the social and ecological fields, but much literature has investigated several of these topics without specifying the occurrence or the implication for these protected areas. Thus, research should capitalise these studies in the new regulatory context.
- A better understanding of the effects that main anthropogenic actions (e.g., human infrastructures, common management practices) have on specific species and habitats by considering their variability in terms of intensity, frequency and cumulative impacts.
- A more comprehensive knowledge of the process and the effects of forest spontaneous development at the global scale. In particular research should focus on expansion rates, affected habitats, multi-scale changes, and biodiversity and ecosystem services implications. Meta-analysis of published data should be prioritized.
- A better understanding of the habitat functions of the Directive’s habitat types for species of conservation interest. While, there is still the need for ecological research on habitat-species relationships, especially in managed contexts, it is also important to simply link animal species to these habitat types to understand the possible conservation implications.
- A focus on landscape patterns analysis of habitat thresholds at different spatial scales for species of conservation interest among differently managed landscapes. Research should focus on comparisons among landscapes representing single management approaches (e.g., abandoned/wilderness, managed) and their different possible integration.
- Model and forecast the potential spread of invasive alien tree species using land abandonment scenarios.
- The investigation of the habitat function, and in particular for species of conservation interest, of novel forest communities developing in urban settings.

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10. References

Angelstam, P., Bütler, R., Lazdinis, M., Mikusiński, G., Roberge, J.-M., 2003. Habitat thresholds for focal species at multiple scales and forest biodiversity conservation — dead wood as an example. Annales Zoologici Fennici 40, 473-482.

Ascoli, D., Lonati, M., Marzano, R., Bovio, G., Cavallero, A., Lombardi, G., 2013. Prescribed burning and browsing to control tree encroachment in southern European heathlands. Forest Ecology and Management 289, 69-77.

Ausden, M., 2007. Habitat management for conservation: a handbook of techniques. Oxford University Press, Oxford.

Barbati, A., Salvati, R., Ferrari, B., Di Santo, D., Quatrini, A., Portoghesi, L., Travaglini, D., Iovino, F., Nocentini, S., 2012. Assessing and promoting old-growthness of forest stands: lessons from research in Italy. Plant Biosystems 146, 167-174.

Barbier, S., Gosselin, F., Balandier, P., 2008. Influence of tree species on understory vegetation diversity and mechanisms involved—A critical review for temperate and boreal forests. Forest Ecology and Management 254, 1-15.

Barlow, J., Gardner, T.A., Araujo, I.S., Ávila-Pires, T.C., Bonaldo, A.B., Costa, J.E., Esposito, M.C., Ferreira, L.V., Hawes, J., Hernandez, M.I.M., Hoogmoed, M.S., Leite, R.N., Lo-Man-Hung, N.F., Malcolm, J.R., Martins, M.B., Mestre, L.A.M., Miranda-Santos, R., Nunes-Gutjahr, A.L., Overal, W.L., Parry, L., Peters, S.L., Ribeiro-Junior, M.A., da Silva, M.N.F., da Silva Motta, C., Peres, C.A., 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. Proceedings of the National Academy of Sciences 104, 18555-18560.

Bauhus, J., Puettmann, K., Messier, C., 2009. Silviculture for old-growth attributes. Forest Ecology and Management 258, 525-537.

Bengtsson, J., Nilsson, S.G., Franc, A., Menozzi, P., 2000. Biodiversity, disturbances, ecosystem function and management of European forests. Forest Ecology and Management 132, 39-50.

Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends in Ecology & Evolution 18, 182-188.

Bergeron, Y., Harvey, B., Leduc, A., Gauthier, S., 1999. Forest management guidelines based on natural disturbance dynamics: stand- and forest-level considerations. The Forestry Chronicle 75, 49-54.

Biondi, E., Burrascano, S., Casavecchia, S., Copiz, R., Del Vico, E., Galdenzi, D., Gigante, D., Lasen, C., Spampinato, G., Venanzoni, R., Zivkovic, L., Blasi, C., 2012. Diagnosis and syntaxonomic interpretation of Annex I Habitats (Dir . 92 / 43 / EEC) in Italy at the alliance level. Plant Sociology 49, 5-37.

Blicharska, M., Orlikowska, E.H., Roberge, J.-M., Grodzinska-Jurczak, M., 2016. Contribution of social science to large scale biodiversity conservation: a review of research about the Natura 2000 network. Biological Conservation 199, 110-122.

Bonan, G.B., Shugart, H.H., 1989. Environmental factors and ecological processes in boreal forests. Annual Review of Ecology and Systematics 20, 1-28.

Bowen, M.E., McAlpine, C.A., House, A.P.N., Smith, G.C., 2007. Regrowth forests on abandoned agricultural land: a review of their habitat values for recovering forest fauna. Biological Conservation 140, 273-296.

Branquart, E., Verheyen, K., Latham, J., 2008. Selection criteria of protected forest areas in Europe: the theory and the real world. Biological Conservation 141, 2795-2806.

Brooks, T.M., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A.B., Rylands, A.B., Konstant, W.R., Flick, P., Pilgrim, J., Oldfield, S., Magin, G., Hilton-Taylor, C., 2002. Habitat loss and extinction in the hotspots of biodiversity. Conservation Biology 16, 909-923.

Brunet, J., Fritz, Ö., Richnau, G., 2010. Biodiversity in European beech forests – a review with recommendations for sustainable forest management. Ecological Bulletins 53, 77-94.

Bunce, R.G.H., Bogers, M.M.B., Evans, D., Halada, L., Jongman, R.H.G., Mucher, C.A., Bauch, B., de Blust, G., Parr, T.W., Olsvig-Whittaker, L., 2013. The significance of habitats as indicators of biodiversity and their links to species. Ecological Indicators 33, 19-25.

Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J.-F., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Morcillo, M.H., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vié, J.-C., Watson, R., 2010. Global biodiversity: indicators of recent declines. Science 328, 1164-1168.

Camenen, E., Porté, A.J., Benito Garzón, M., 2016. American trees shift their niches when invading Western Europe: evaluating invasion risks in a changing climate. Ecology and Evolution 6, 7263-7275.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., Naeem, S., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59-67.

Ceauşu, S., Carver, S., Verburg, P.H., Kuechly, H.U., Hölker, F., Brotons, L., Pereira, H.M., 2015. European wilderness in a time of farmland abandonment. In: Pereira, H.M., Navarro, L.M. (Eds.), Rewilding European landscapes. Springer International Publishing, Cham, pp. 25-46.

Chamberlain, D.E., Wilson, A.M., Browne, S.J., Vickery, J.A., 1999. Effects of habitat type and management on the abundance of skylarks in the breeding season. Journal of Applied Ecology 36, 856-870.

Chazdon, R.L., 2008. Beyond deforestation: restoring forests and ecosystem services on degraded lands. Science 320, 1458-1460.

Chirici, G., Winter, S., McRoberts, R.E., 2011. National forest inventories: contributions to forest biodiversity assessments. Springer, Heidelberg.

Chylarecki, P., Selva, N., 2016. Ancient forest: spare it from clearance. Nature 530, 419.

Cousins, S.A.O., Eriksson, O., 2002. The influence of management history and habitat on plant species richness in a rural hemiboreal landscape, Sweden. Landscape Ecology 17, 517-529.

CPF, 2008. Strategic framework for forests and climate change. A proposal by the Collaborative Partnership on Forests for a coordinated forest-sector response to climate change. The Collaborative Partnership on Forests, New York.

Cullotta, S., Bončina, A., Carvalho-Ribeiro, S.M., Chauvin, C., Farcy, C., Kurttila, M., Maetzke, F.G., 2015. Forest planning across Europe: the spatial scale, tools, and inter-sectoral integration in land-use planning. Journal of Environmental Planning and Management 58, 1384-1411.

DAISIE, 2009. Handbook of alien species in Europe. Springer, Dordrecht.

Daubenmire, R., 1968. Plant communities. Harper and Row, New York.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity 7, 260-272.

Denevan, W.M., 1992. The pristine myth: the landscape of the Americas in 1492. Annals of the Association of American Geographers 82, 369-385.

Dent, D.H., Joseph Wright, S., 2009. The future of tropical species in secondary forests: a quantitative review. Biological Conservation 142, 2833-2843.

Diamond, J.M., 1989. Overview of recent extinctions. In: Western, D., Pearl, M.C. (Eds.), Conservation for the twenty-first century. Oxford Univeristy Press, Oxford, pp. 37-41.

Dickie, I.A., Bennett, B.M., Burrows, L.E., Nuñez, M.A., Peltzer, D.A., Porté, A., Richardson, D.M., Rejmánek, M., Rundel, P.W., van Wilgen, B.W., 2014. Conflicting values: ecosystem services and invasive tree management. Biological Invasions 16, 705-719.

Didham, R.K., Tylianakis, J.M., Gemmell, N.J., Rand, T.A., Ewers, R.M., 2007. Interactive effects of habitat modification and species invasion on native species decline. Trends in Ecology & Evolution 22, 489-496.

Doak, D.F., Marino, P.C., Kareiva, P.M., 1992. Spatial scale mediates the influence of habitat fragmentation on dispersal success: implications for conservation. Theoretical Population Biology 41, 315-336.

Duncker, P.S., Barreiro, S.M., Hengeveld, G.M., Lind, T., Mason, W.L., Ambrozy, S., Spiecker, H., 2012. Classification of forest management approaches: a new conceptual framework and its applicability to European forestry. Ecology and Society 17, 51.

Dupouey, J.L., Dambrine, E., Laffite, J.D., Moares, C., 2002. Irreversible impact of past land use on forest soils and biodiversity. Ecology 83, 2978-2984.

Early, R., Bradley, B.A., Dukes, J.S., Lawler, J.J., Olden, J.D., Blumenthal, D.M., Gonzalez, P., Grosholz, E.D., Ibañez, I., Miller, L.P., Sorte, C.J.B., Tatem, A.J., 2016. Global threats from invasive alien species in the twenty-first century and national response capacities. Nature Communications 7, 12485.

EC, 2013. Interpretation manual of European Union habitats - EU 28. European Commission, Luxembourg.

EEA, 2015. State of nature in the EU. Results from reporting under the nature directives 2007-2012. European Environmental Agency, Copenhagen.

EEA, 2016. European forest ecosystems. State and trends. Publications Office of the European Union, Luxembourg.

Ervin, J., 2003. Rapid assessment of protected area management effectiveness in four countries. BioScience 53, 833-841.

Evans, D., 2006. The habitats of the European Union Habitats Directive. Biology and Environment: Proceedings of the Royal Irish Academy 106B, 167-173.

Evans, D., 2010. Interpreting the habitats of Annex I: past, present and future. Acta Botanica Gallica 157, 677-686.

Evans, D., 2012. Building the European Union’s Natura 2000 network. Nature Conservation 1, 11-26.

FAO, 2014. State of the world's forests 2014: enhancing the socioeconomic benefits from forests. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2016. Global forest resources assessment. How are the world's forests changing? Food and Agriculture Organization of the United Nations, Rome.

Feinerer, I., Hornik, K., 2015. tm: Text Mining Package version 0.6-2.

Fellows, I., 2013. Wordcloud: Word Clouds. R package version 2.4.

Ferraro, P.J., Pattanayak, S.K., 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biology 4, e105.

Forest Europe, 2015. State of Europe's Forests. Ministerial Conference on the Protection of Forests in Europe, Madrid.

Gálhidy, L., Mihók, B., Hagyó, A., Rajkai, K., Standovár, T., 2006. Effects of gap size and associated changes in light and soil moisture on the understorey vegetation of a Hungarian beech forest. Plant Ecology 183, 133-145.

Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A., Sodhi, N.S., 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecology Letters 12, 561-582.

Gaspar, C., Gaston, K.J., Borges, P.A.V., Cardoso, P., 2011. Selection of priority areas for arthropod conservation in the Azores archipelago. Journal of Insect Conservation 15, 671-684.

Geijzendorffer, I.R., Regan, E.C., Pereira, H.M., Brotons, L., Brummitt, N., Gavish, Y., Haase, P., Martin, C.S., Mihoub, J.-B., Secades, C., Schmeller, D.S., Stoll, S., Wetzel, F.T., Walters, M., 2016. Bridging the gap between biodiversity data and policy reporting needs: an essential biodiversity variables perspective. Journal of Applied Ecology 53, 1341-1350.

Gibson, L., Lee, T.M., Koh, L.P., Brook, B.W., Gardner, T.A., Barlow, J., Peres, C.A., Bradshaw, C.J.A., Laurance, W.F., Lovejoy, T.E., Sodhi, N.S., 2011. Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478, 378-381.

Gómez-Pompa, A., Kaus, A., 1992. Taming the wilderness myth. BioScience 42, 271-279.

Götmark, F., 2013. Habitat management alternatives for conservation forests in the temperate zone: review, synthesis, and implications. Forest Ecology and Management 306, 292-307.

Grantham, H.S., Pressey, R.L., Wells, J.A., Beattie, A.J., 2010. Effectiveness of biodiversity surrogates for conservation planning: different measures of effectiveness generate a kaleidoscope of variation. PLoS ONE 5, e11430.

Grindal, S.D., Brigham, R.M., 1999. Impacts of forest harvesting on habitat use by foraging insectivorous bats at different spatial scales. Ecoscience 6, 25-34.

Guisan, A., Thuiller, W., 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters 8, 993-1009.

Guisan, A., Zimmermann, N.E., 2000. Predictive habitat distribution models in ecology. Ecological Modelling 135, 147-186.

Gustafsson, L., Baker, S.C., Bauhus, J., Beese, W.J., Brodie, A., Kouki, J., Lindenmayer, D.B., Lõhmus, A., Pastur, G.M., Messier, C., Neyland, M., Palik, B., Sverdrup-Thygeson, A., Volney, W.J.A., Wayne, A., Franklin, J.F., 2012. Retention forestry to maintain multifunctional forests: a world perspective. BioScience 62, 633-645.

Haddaway, N.R., Styles, D., Pullin, A.S., 2014. Evidence on the environmental impacts of farm land abandonment in high altitude/mountain regions: a systematic map. Environmental Evidence 3, 17.

Halada, L., Evans, D., Romão, C., Petersen, J.-E., 2011. Which habitats of European importance depend on agricultural practices? Biodiversity and Conservation 20, 2365-2378.

Hall, L.S., Krausman, P.R., Morrison, M.L., 1997. The habitat concept and a plea for standard terminology. Wildlife Society Bulletin 25, 173-182.

Halpin, P.N., 1997. Global climate change and natural-area protection: management responses and research directions. Ecological Applications 7, 828-843.

Hobbs, R.J., Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., Epstein, P.R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W., Valladares, F., Vilà, M., Zamora, R., Zobel, M., 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15, 1-7.

Hobbs, R.J., Cramer, V.A., 2007. Why old fields? Socioeconomic and ecological causes and consequences of land abandonment. In: Cramer, V.A., Hobbs, R.J. (Eds.), Old fields: dynamics and restoration of abandoned farmland. Island Press, Washington DC.

Höchtl, F., Lehringer, S., Konold, W., 2005. “Wilderness”: what it means when it becomes a reality—a case study from the southwestern Alps. Landscape and Urban Planning 70, 85-95.

Hockings, M., 2003. Systems for assessing the effectiveness of management in protected areas. BioScience 53, 823-832.

Hole, D.G., Willis, S.G., Pain, D.J., Fishpool, L.D., Butchart, S.H.M., Collingham, Y.C., Rahbek, C., Huntley, B., 2009. Projected impacts of climate change on a continent-wide protected area network. Ecology Letters 12, 420-431.

Joppa, L.N., Loarie, S.R., Pimm, S.L., 2008. On the protection of “protected areas”. Proceedings of the National Academy of Sciences 105, 6673-6678.

Joppa, L.N., Pfaff, A., 2010. Global protected area impacts. Proceedings of the Royal Society B: Biological Sciences.

Kearney, M., 2006. Habitat, environment and niche: what are we modelling? Oikos 115, 186-191.

Keeton, W.S., 2006. Managing for late-successional/old-growth characteristics in northern hardwood-conifer forests. Forest Ecology and Management 235, 129-142.

Kindlmann, P., Krenova, Z., 2016. Biodiversity: Protect Czech park from development. Nature 531, 448-448.

Kowarik, I., 2011. Novel urban ecosystems, biodiversity, and conservation. Environmental Pollution 159, 1974-1983.

Kuuluvainen, T., 2009. Forest management and biodiversity conservation based on natural ecosystem dynamics in northern Europe: the complexity challenge. Ambio 38, 309-315.

Lamarque, L.J., Delzon, S., Lortie, C.J., 2011. Tree invasions: a comparative test of the dominant hypotheses and functional traits. Biological Invasions 13, 1969-1989.

Lasanta, T., Arnáez, J., Pascual, N., Ruiz-Flaño, P., Errea, M.P., Lana-Renault, N., 2017. Space–time process and drivers of land abandonment in Europe. Catena 149, 810-823.

Lasanta, T., Nadal-Romero, E., Arnáez, J., 2015. Managing abandoned farmland to control the impact of re-vegetation on the environment. The state of the art in Europe. Environmental Science & Policy 52, 99-109.

Law, B.S., Dickman, C.R., 1998. The use of habitat mosaics by terrestrial vertebrate fauna: implications for conservation and management. Biodiversity & Conservation 7, 323-333.

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T.M., Bertzky, B., Butchart, S.H.M., Stuart, S.N., Badman, T., Rodrigues, A.S.L., 2013. Protected areas and effective biodiversity conservation. Science 342, 803-805.

Leverington, F., Costa, K.L., Pavese, H., Lisle, A., Hockings, M., 2010. A global analysis of protected area management effectiveness. Environmental Management 46, 685-698.

Liang, J., Crowther, T.W., Picard, N., Wiser, S., Zhou, M., Alberti, G., Schulze, E.-D., McGuire, A.D., Bozzato, F., Pretzsch, H., de-Miguel, S., Paquette, A., Hérault, B., Scherer-Lorenzen, M., Barrett, C.B., Glick, H.B., Hengeveld, G.M., Nabuurs, G.-J., Pfautsch, S., Viana, H., Vibrans, A.C., Ammer, C., Schall, P., Verbyla, D., Tchebakova, N., Fischer, M., Watson, J.V., Chen, H.Y.H., Lei, X., Schelhaas, M.-J., Lu, H., Gianelle, D., Parfenova, E.I., Salas, C., Lee, E., Lee, B., Kim, H.S., Bruelheide, H., Coomes, D.A., Piotto, D., Sunderland, T., Schmid, B., Gourlet-Fleury, S., Sonké, B., Tavani, R., Zhu, J., Brandl, S., Vayreda, J., Kitahara, F., Searle, E.B., Neldner, V.J., Ngugi, M.R., Baraloto, C., Frizzera, L., Bałazy, R., Oleksyn, J., Zawiła-Niedźwiecki, T., Bouriaud, O., Bussotti, F., Finér, L., Jaroszewicz, B., Jucker, T., Valladares, F., Jagodzinski, A.M., Peri, P.L., Gonmadje, C., Marthy, W., O'Brien, T., Martin, E.H., Marshall, A.R., Rovero, F., Bitariho, R., Niklaus, P.A., Alvarez-Loayza, P., Chamuya, N., Valencia, R., Mortier, F., Wortel, V., Engone-Obiang, N.L., Ferreira, L.V., Odeke, D.E., Vasquez, R.M., Lewis, S.L., Reich, P.B., 2016. Positive biodiversity-productivity relationship predominant in global forests. Science 354, aaf8957.

Lindbladh, M., Bradshaw, R., 1998. The origin of present forest composition and pattern in southern Sweden. Journal of Biogeography 25, 463-477.

Lindenmayer, D., McCarthy, M.A., 2002. Congruence between natural and human forest disturbance: a case study from Australian montane ash forests. Forest Ecology and Management 155, 319-335.

Lindenmayer, D.B., 1999. Future directions for biodiversity conservation in managed forests: indicator species, impact studies and monitoring programs. Forest Ecology and Management 115, 277-287.

Lindenmayer, D.B., 2000. Factors at multiple scales affecting distribution patterns and their implications for animal conservation – Leadbeater's Possum as a case study. Biodiversity & Conservation 9, 15-35.

Lindenmayer, D.B., Franklin, J.F., 2002. Conserving forest biodiversity: a comprehensive multiscale approach. Island Press, Washington.

Lindenmayer, D.B., Franklin, J.F., Fischer, J., 2006. General management principles and a checklist of strategies to guide forest biodiversity conservation. Biological Conservation 131, 433-445.

Lindenmayer, D.B., Margules, C.R., Botkin, D.B., 2000. Indicators of biodiversity for ecologically sustainable forest management. Conservation Biology 14, 941-950.

Long, J.N., 2009. Emulating natural disturbance regimes as a basis for forest management: a North American view. Forest Ecology and Management 257, 1868-1873.

MacDonald, D., Crabtree, J., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., Gibon, A., 2000. Agricultural abandonment in mountain areas of Europe: environmental consequences and policy response. Journal of Environmental Management 59, 47-69.

MacDougall, A.S., Turkington, R., 2005. Are invasive alien species the drivers or passengers of change in degraded ecosystems? Ecology 86, 42-55.

Margules, C.R., Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243-253.

McCollin, D., Geraghty, E., 2015. Change and causes of change in the vascular plant flora of Ireland: 1970–1999. Proceedings of the Royal Irish Academy 115B, 67-81.

McCollin, D., Moore, L., Sparks, T.H., 2000. The flora of a cultural landscape: environmental determinants of change revealed using archival sources. Biological Conservation 92, 249-263.

McCollin, D., Preece, R.C., Sparks, T.H., 2015. Pushing back the baseline: a novel approach to detect long-term changes in terrestrial faunal abundance using historical qualitative descriptions. European Journal of Ecology 1, 32-42.

Meyfroidt, P., Lambin, E.F., 2011. Global forest transition: prospects for an end to deforestation. Annual Review of Environment and Resources 36, 343-371.

Miller, C.J., 2000. Vegetation and habitat are not synonyms. Ecological Management & Restoration 1, 102-104.

Miller, J.R., Hobbs, R.J., 2007. Habitat restoration—Do we know what we're doing? Restoration Ecology 15, 382-390.

Mitchell, S.C., 2005. How useful is the concept of habitat? – a critique. Oikos 110, 634-638.

Mori, A.S., Kitagawa, R., 2014. Retention forestry as a major paradigm for safeguarding forest biodiversity in productive landscapes: a global meta-analysis. Biological Conservation 175, 65-73.

Mori, A.S., Lertzman, K.P., Gustafsson, L., 2017. Biodiversity and ecosystem services in forest ecosystems: a research agenda for applied forest ecology. Journal of Applied Ecology 54, 12-27.

Morrison, M.L., Marcot, B.G., Mannan, R.W., 2006. Wildlife-habitat relationships: concepts and applications. Island Press, Washington, Covelo, London.

Müller, J., Bütler, R., 2010. A review of habitat thresholds for dead wood: a baseline for management recommendations in European forests. European Journal of Forest Research 129, 981-992.

Müllerová, J., Szabó, P., Hédl, R., 2014. The rise and fall of traditional forest management in southern Moravia: a history of the past 700 years. Forest Ecology and Management 331, 104-115.

Munteanu, C., Nita, M.D., Abrudan, I.V., Radeloff, V.C., 2016. Historical forest management in Romania is imposing strong legacies on contemporary forests and their management. Forest Ecology and Management 361, 179-193.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Naughton-Treves, L., Holland, M.B., Brandon, K., 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. Annual Review of Environment and Resources 30, 219-252.

Navarro, L.M., Pereira, H.M., 2012. Rewilding abandoned landscapes in Europe. Ecosystems 15, 900-912.

New, T.R., Pyle, R.M., Thomas, J.A., Thomas, C.D., Hammond, P.C., 1995. Butterfly conservation management. Annual Review of Entomology 40, 57-83.

Newbold, T., Scharlemann, J.P.W., Butchart, S.H.M., Şekercioğlu, Ç.H., Joppa, L., Alkemade, R., Purves, D.W., 2014. Functional traits, land-use change and the structure of present and future bird communities in tropical forests. Global Ecology and Biogeography 23, 1073-1084.

Nogués-Bravo, D., Simberloff, D., Rahbek, C., Sanders, N.J., 2016. Rewilding is the new Pandora's box in conservation. Current Biology 26, R87-R91.

Noss, R.F., 1999. Assessing and monitoring forest biodiversity: a suggested framework and indicators. Forest Ecology and Management 115, 135-146.

O'Hara, K.L., 2001. The silviculture of transformation — a commentary. Forest Ecology and Management 151, 81-86.

Opdam, P., Wascher, D., 2004. Climate change meets habitat fragmentation: linking landscape and biogeographical scale levels in research and conservation. Biological Conservation 117, 285-297.

Orlikowska, E.H., Roberge, J.-M., Blicharska, M., Mikusiński, G., 2016. Gaps in ecological research on the world's largest internationally coordinated network of protected areas: a review of Natura 2000. Biological Conservation 200, 216-227.

Ostermann, O.P., 1998. The need for management of nature conservation sites designated under Natura 2000. Journal of Applied Ecology 35, 968-973.

Paillet, Y., Bergès, L., Hjältén, J., Ódor, P., Avon, C., Bernhardt-Römermann, M., Bijlsma, R.-J., De Bruyn, L.U.C., Fuhr, M., Grandin, U.L.F., Kanka, R., Lundin, L., Luque, S., Magura, T., Matesanz, S., Mészáros, I., Sebastiá, M.T., Schmidt, W., StandováR, T., Tóthmérész, B., Uotila, A., Valladares, F., Vellak, K.A.I., Virtanen, R., 2010. Biodiversity differences between managed and unmanaged forests: meta-analysis of species richness in Europe. Conservation Biology 24, 101-112.

Pascual-Hortal, L., Saura, S., 2007. Impact of spatial scale on the identification of critical habitat patches for the maintenance of landscape connectivity. Landscape and Urban Planning 83, 176-186.

Pereira, H.M., Navarro, L.M., 2015. Rewilding European landscapes. Springer, Heidelberg.

Pergl, J., Genovesi, P., Pysek, P., 2016. Europe: better management of alien species. Nature 531, 173.

Peterken, G.F., 1981. Woodland Conservation and Management. Chapman & Hall, London and New York.

Peters, D.M., Schraml, U., 2014. Does background matter? Disciplinary perspectives on sustainable forest management. Biodiversity and Conservation 23, 3373-3389.

Petit, R.J., Bialozyt, R., Garnier-Géré, P., Hampe, A., 2004. Ecology and genetics of tree invasions: from recent introductions to Quaternary migrations. Forest Ecology and Management 197, 117-137.

Pimm, S.L., Ayres, M., Balmford, A., Branch, G., Brandon, K., Brooks, T., Bustamante, R., Costanza, R., Cowling, R., Curran, L.M., Dobson, A., Farber, S., da Fonseca, G.A.B., Gascon, C., Kitching, R., McNeely, J., Lovejoy, T., Mittermeier, R.A., Myers, N., Patz, J.A., Raffle, B., Rapport, D., Raven, P., Roberts, C., Rodríguez, J.P., Rylands, A.B., Tucker, C., Safina, C., Samper, C., Stiassny, M.L.J., Supriatna, J., Wall, D.H., Wilcove, D., 2001. Can we defy nature's end? Science 293, 2207-2208.

Pommerening, A., Murphy, S.T., 2004. A review of the history, definitions and methods of continuous cover forestry with special attention to afforestation and restocking. Forestry 77, 27-44.

Puettmann, K.J., Ammer, C., 2007. Trends in North American and European regeneration research under the ecosystem management paradigm. European Journal of Forest Research 126, 1-9.

Puettmann, K.J., Coates, K.D., Messier, C., 2009. A critique to silviculture: managing complexity. Island Press, Washington, D.C.

Pyšek, P., Jarošík, V., Hulme, P.E., Pergl, J., Hejda, M., Schaffner, U., Vilà, M., 2012. A global assessment of invasive plant impacts on resident species, communities and ecosystems: the interaction of impact measures, invading species' traits and environment. Global Change Biology 18, 1725-1737.

Queiroz, C., Beilin, R., Folke, C., Lindborg, R., 2014. Farmland abandonment: threat or opportunity for biodiversity conservation? A global review. Frontiers in Ecology and the Environment 12, 288-296.

R Development Core Team, 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Rackham, O., 1976. Trees and woodland in the British landscape. J. M. Dent & Sons Ltd., London.

Rackham, O., 1980. Ancient woodland, its history, vegetation and uses in England. Edward Arnold, London.

Rackham, O., 1986. The history of the countryside: the full fascinating story of Britain's landscape. J. M. Dent & Sons Ltd., London.

Rackham, O., 2008. Ancient woodlands: modern threats. New Phytologist 180, 571-586.

Rametsteiner, E., Simula, M., 2003. Forest certification—an instrument to promote sustainable forest management? Journal of Environmental Management 67, 87-98.

Rands, M.R.W., Adams, W.M., Bennun, L., Butchart, S.H.M., Clements, A., Coomes, D., Entwistle, A., Hodge, I., Kapos, V., Scharlemann, J.P.W., Sutherland, W.J., Vira, B., 2010. Biodiversity conservation: challenges beyond 2010. Science 329, 1298-1303.

Rey Benayas, J., Martins, A., Nicolau, J.M., Schulz, J.J., 2007. Abandonment of agricultural land: an overview of drivers and consequences. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources 2, 1-14.

Rey Benayas, J.M., Bullock, J.M., 2012. Restoration of biodiversity and ecosystem services on agricultural land. Ecosystems 15, 883-899.

Richardson, D.M., 1998. Forestry trees as invasive aliens. Conservation Biology 12, 18-26.

Richardson, D.M., Pyšek, P., Rejmánek, M., Barbour, M.G., Panetta, F.D., West, C.J., 2000. Naturalization and invasion of alien plants: concepts and definitions. Diversity and Distributions 6, 93-107.

Richardson, D.M., Rejmánek, M., 2011. Trees and shrubs as invasive alien species – a global review. Diversity and Distributions 17, 788-809.

Roberts, M.R., Gilliam, F.S., 2003. Response of the herbaceous layer to disturbance in eastern forests. In: Gilliam, F.S., Roberts, M.R. (Eds.), The herbaceous layer in forests of eastern North America. Oxford University Press, New York, pp. 302-320.

Rondinini, C., Di Marco, M., Chiozza, F., Santulli, G., Baisero, D., Visconti, P., Hoffmann, M., Schipper, J., Stuart, S.N., Tognelli, M.F., Amori, G., Falcucci, A., Maiorano, L., Boitani, L., 2011. Global habitat suitability models of terrestrial mammals. Philosophical Transactions of the Royal Society B: Biological Sciences 366, 2633-2641.

Rowe, J.S., Scotter, G.W., 1973. Fire in the boreal forest. Quaternary Research 3, 444-464.

Rudel, T.K., Coomes, O.T., Moran, E., Achard, F., Angelsen, A., Xu, J., Lambin, E., 2005. Forest transitions: towards a global understanding of land use change. Global Environmental Change 15, 23-31.

Rudel, T.K., Defries, R., Asner, G.P., Laurance, W.F., 2009. Changing Drivers of Deforestation and New Opportunities for Conservation. Conservation Biology 23, 1396-1405.

Sala, O.E., Stuart Chapin , F., III, Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M.n., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770-1774.

Sarkar, S., 1999. Wilderness preservation and biodiversity conservation—keeping divergent goals distinct. BioScience 49, 405-412.

Schirmel, J., Bundschuh, M., Entling, M.H., Kowarik, I., Buchholz, S., 2016. Impacts of invasive plants on resident animals across ecosystems, taxa, and feeding types: a global assessment. Global Change Biology 22, 594-603.

Schroeder, L.M., Ranius, T., Ekbom, B., Larsson, S., 2006. Recruitment of saproxylic beetles in high stumps created for maintaining biodiversity in a boreal forest landscape. Canadian Journal of Forest Research 36, 2168-2178.

Selva, N., 2016. Defining forest, habitat, and wilderness in a changing world. Conservation Biology 30, 1140-1142.

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D.A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., Vilà, M., 2013. Impacts of biological invasions: what's what and the way forward. Trends in Ecology & Evolution 28, 58-66.

Sitzia, T., Campagnaro, T., Dainese, M., Cierjacks, A., 2012. Plant species diversity in alien black locust stands: a paired comparison with native stands across a north-Mediterranean range expansion. Forest Ecology and Management 285, 85-91.

Sitzia, T., Semenzato, P., Trentanovi, G., 2010. Natural reforestation is changing spatial patterns of rural mountain and hill landscapes: a global overview. Forest Ecology and Management 259, 1354-1362.

Sjöman, H., Morgenroth, J., Sjöman, J.D., Sæbø, A., Kowarik, I., 2016. Diversification of the urban forest—Can we afford to exclude exotic tree species? Urban Forestry & Urban Greening 18, 237-241.

Soulé, M., Noss, R., 1998. Rewilding and biodiversity: complementary goals for continental conservation. Wild Earth 8, 18-28.

Suding, K.N., Hobbs, R.J., 2009. Threshold models in restoration and conservation: a developing framework. Trends in Ecology & Evolution 24, 271-279.

Svenning, J.-C., Pedersen, P.B.M., Donlan, C.J., Ejrnæs, R., Faurby, S., Galetti, M., Hansen, D.M., Sandel, B., Sandom, C.J., Terborgh, J.W., Vera, F.W.M., 2016. Science for a wilder Anthropocene: synthesis and future directions for trophic rewilding research. Proceedings of the National Academy of Sciences 113, 898-906.

Szabó, P., Müllerová, J., Suchánková, S., Kotačka, M., 2015. Intensive woodland management in the Middle Ages: spatial modelling based on archival data. Journal of Historical Geography 48, 1-10.

Thomas, C.D., Cameron, A., Green, R.E., Bakkenes, M., Beaumont, L.J., Collingham, Y.C., Erasmus, B.F.N., de Siqueira, M.F., Grainger, A., Hannah, L., Hughes, L., Huntley, B., van Jaarsveld, A.S., Midgley, G.F., Miles, L., Ortega-Huerta, M.A., Townsend Peterson, A., Phillips, O.L., Williams, S.E., 2004. Extinction risk from climate change. Nature 427, 145-148.

Tittensor, D.P., Walpole, M., Hill, S.L.L., Boyce, D.G., Britten, G.L., Burgess, N.D., Butchart, S.H.M., Leadley, P.W., Regan, E.C., Alkemade, R., Baumung, R., Bellard, C., Bouwman, L., Bowles-Newark, N.J., Chenery, A.M., Cheung, W.W.L., Christensen, V., Cooper, H.D., Crowther, A.R., Dixon, M.J.R., Galli, A., Gaveau, V., Gregory, R.D., Gutierrez, N.L., Hirsch, T.L., Höft, R., Januchowski-Hartley, S.R., Karmann, M., Krug, C.B., Leverington, F.J., Loh, J., Lojenga, R.K., Malsch, K., Marques, A., Morgan, D.H.W., Mumby, P.J., Newbold, T., Noonan-Mooney, K., Pagad, S.N., Parks, B.C., Pereira, H.M., Robertson, T., Rondinini, C., Santini, L., Scharlemann, J.P.W., Schindler, S., Sumaila, U.R., Teh, L.S.L., van Kolck, J., Visconti, P., Ye, Y., 2014. A mid-term analysis of progress toward international biodiversity targets. Science 346, 241-244.

Tomaselli, V., Dimopoulos, P., Marangi, C., Kallimanis, A.S., Adamo, M., Tarantino, C., Panitsa, M., Terzi, M., Veronico, G., Lovergine, F., Nagendra, H., Lucas, R., Mairota, P., Mücher, C.A., Blonda, P., 2013. Translating land cover/land use classifications to habitat taxonomies for landscape monitoring: a Mediterranean assessment. Landscape Ecology 28, 905-930.

Trentanovi, G., von der Lippe, M., Sitzia, T., Ziechmann, U., Kowarik, I., Cierjacks, A., 2013. Biotic homogenization at the community scale: disentangling the roles of urbanization and plant invasion. Diversity and Distributions 19, 738-748.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. Ecology Letters 8, 857-874.

Tsiafouli, M.A., Apostolopoulou, E., Mazaris, A.D., Kallimanis, A.S., Drakou, E.G., Pantis, J.D., 2013. Human activities in Natura 2000 sites: a highly diversified conservation network. Environmental Management 51, 1025-1033.

Turner, N.J., 2001. "Doing it right": issues and practices of sustainable harvesting of non-timber forest products relating to First Peoples in British Columbia.

van Vliet, J., Magliocca, N.R., Büchner, B., Cook, E., Rey Benayas, J.M., Ellis, E.C., Heinimann, A., Keys, E., Lee, T.M., Liu, J., Mertz, O., Meyfroidt, P., Moritz, M., Poeplau, C., Robinson, B.E., Seppelt, R., Seto, K.C., Verburg, P.H., 2016. Meta-studies in land use science: current coverage and prospects. Ambio 45, 15-28.

Vilà, M., Basnou, C., Pyšek, P., Josefsson, M., Genovesi, P., Gollasch, S., Nentwig, W., Olenin, S., Roques, A., Roy, D., Hulme, P.E., partners, D., 2010. How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. Frontiers in Ecology and the Environment 8, 135-144.

Vilà, M., Espinar, J.L., Hejda, M., Hulme, P.E., Jarošík, V., Maron, J.L., Pergl, J., Schaffner, U., Sun, Y., Pyšek, P., 2011. Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities and ecosystems. Ecology Letters 14, 702-708.

Webb, N.R., 1998. The traditional management of European heathlands. Journal of Applied Ecology 35, 987-990.

Whitney, G.G., Davis, W.C., 1986. From primitive woods to cultivated woodlots: Thoreau and the forest history of Concord, Massachusetts. Forest & Conservation History 30, 70-81.

Whittaker, R.H., Levin, S.A., Root, R.B., 1973. Niche, habitat, and ecotope. The American Naturalist 107, 321-338.

Wiersum, K.F., 1995. 200 years of sustainability in forestry: lessons from history. Environmental Management 19, 321-329.

Winkel, G., Jump, A., 2014. Perspectives on forest conservation: building evidence at the frontier between policy and conservation science. Biodiversity and Conservation 23, 3359-3372.

Zurlini, G., Riitters, K., Zaccarelli, N., Petrosillo, I., Jones, K.B., Rossi, L., 2006. Disturbance patterns in a socio-ecological system at multiple scales. Ecological Complexity 3, 119-128.