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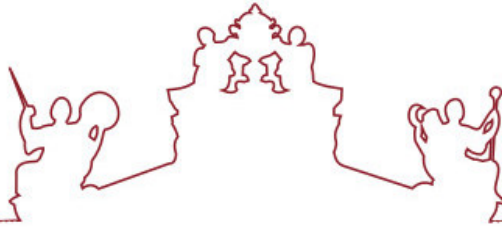Programa de Doutoramento em Ciências Agrárias e Ambientais

Tese de Doutoramento

Construction and application of a landscape dynamics assessment tool

Rui Daniel Parreira Machado

Orientador(es) | Janez Pirnat
Nuno Alexandre Neves
Pedro Santos

Évora 2023

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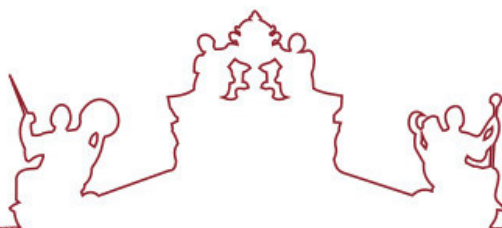Tese de Doutoramento

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CONSTRUCTION AND APPLICATION OF A LANDSCAPE DYNAMICS ASSESSMENT TOOL

ABSTRACT

Landscapes are shaped and influenced by a dynamic balance of ecological processes and anthropogenic activity. Substantial changes may disrupt the natural balance of ecosystems, jeopardizing their capacity to provide essential services for the population. Landscape changes, often measured via land cover variations, are pointed out as a major cause of biodiversity loss and, thus, a central topic in Conservation Biology. Some authors warned that key concepts like 'habitat loss and fragmentation' are often misused, causing misinterpretations and erroneous conclusions. The central aim of this work is to develop an analytical method of assessing landscape dynamics and a conceptual framework capable of clarifying the spatial patterns of land transformation. The conceptual framework, the associated terminology, and the method called Landscape Dynamic Typology, were developed based on the landscape composition and configuration variations. To complement and enhance the methodological approach, two tools were built: i) LDTtool, an ArcGIS toolbox and ii) LDT4QGIS, a group of scripts for QGIS. The tools' multiple features and settings were demonstrated in illustrative examples and case studies as distinct as grasslands preservation in Slovenia, invasive plants removal in a mountainous Natura 2000 site, or the characterization of *montado* loss in Portugal. The use of these software by other authors in urban studies in places like Romania and Indonesia further attests to their usefulness and acceptance by the community. Both tools are freely available and can be adapted to fulfil analysts' needs. New versions will be released whenever a relevant update or improvement is implemented. In specialized or in more comprehensive approaches, complementing or complemented by other software, Landscape Dynamic Typology has shown to be useful in analytical procedures involving land use or land cover changes. By allowing iterative simulations, assisting in change detection and in spatial pattern classification and visualization, it contributes to more informed decision-making.

Keywords

Land Use; Land Cover; Spatial Patterns; Landscape Ecology; GIS; Habitat Fragmentation; Software

CONSTRUÇÃO E APLICAÇÃO DE UMA FERRAMENTA DE AVALIAÇÃO DE DINÂMICAS DE PAISAGENS

RESUMO

As paisagens são moldadas por um equilíbrio dinâmico de processos ecológicos e actividade antropogénica. Alterações substanciais podem perturbar o equilíbrio natural dos ecossistemas, diminuindo a sua capacidade para fornecer serviços essenciais à população. As alterações paisagísticas, geralmente medidas através da variação da ocupação do solo, são uma das principais causas da perda de biodiversidade e um tópico central em biologia da conservação. Alguns autores alertaram que conceitos como 'perda e fragmentação de habitats' são frequentemente mal aplicados, originando interpretações e conclusões erradas. O objectivo central deste trabalho é desenvolver uma estrutura conceptual para clarificar os padrões espaciais de transformação e um construir um método de análise de dinâmicas paisagísticas. O enquadramento conceptual, a terminologia associada e o método, Tipologia de Dinâmicas de Paisagem, foram desenvolvidos com base na variação da composição do mosaico de ocupação do solo. Para complementar a abordagem metodológica foram construídas duas ferramentas: i) LDTtool, uma *toolbox* para ArcGIS; e ii) LDT4QGIS, um grupo de *scripts* para QGIS. As suas funcionalidades foram demonstradas em exemplos e casos de estudo tão díspares como a conservação de prados na Eslovénia, remoção de plantas invasoras num sítio da Rede Natura 2000 em ambiente montanhoso ou a caracterização da perda de áreas de montado em Portugal. O uso deste *software* por outros autores, nomeadamente em estudos urbanos na Roménia e na Indonésia, atesta a sua usabilidade e aceitação pela comunidade. As ferramentas estão disponíveis gratuitamente e podem ser adaptadas para fazer face às necessidades do analista. Serão lançadas novas versões quando novas funcionalidades forem desenvolvidas. Em abordagens especializadas ou generalistas, a Tipologia de Dinâmicas de Paisagem, mostrou ser útil para procedimentos analíticos envolvendo alterações de ocupação do solo. Auxiliando na detecção de alterações, classificação e visualização de padrões espaciais e permitindo fazer simulações, contribui para uma tomada de decisão informada.

Palavras-chave:

Uso do Solo; Ocupação do Solo; Padrões Espaciais; Ecologia da Paisagem; SIG, Fragmentação de Habitat, Software.

ABBREVIATIONS

AEM - Agri-environmental measures

AU - Analytical unit

CLC - CORINE Land Cover

COHESION - Patch Cohesion Index

CRS - Coordinate Reference System

FC – Feature Class

GIS – Geographic Information System

HNV - High Nature Value

IAP - Invasive Alien Plants

IAS - Invasive Alien Species

IIC - Integral Index of Connectivity

LDT - Landscape Dynamics Typology

LPI - Largest Patch Index

LULC - Land Use / Land Cover

MAUP - Modifiable Areal Unit Problem

MESH - Effective Mesh Size

NP - Number of patches

PAR - Perimeter-Area Ratio

PCMM - Patch Corridor Matrix Model

PNG - Pastures and Natural Grasslands

RBS - Result-based schemes

SAC - Special Area of Conservation

SHAPE_MN - Shape Index Distribution (Mean)

ToD - Type of dynamic

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CHAPTER 1. Introduction

1.1. Landscapes, their changes and impacts

The concept of landscape is hard to define due to its complexity and because it allows a wide range of approaches, usually determined by the user's expertise (Cancela D'Abreu et al. 2004). For some, the landscape may represent the amount of territory a person can see from a certain point. Others may add a sense of aesthetics. It is, however, in modern technical approaches, inevitable to include an ecosystemic dimension and to focus on the role of humans in the definition of landscape. Forman (1995) defines landscape as *'a mosaic where the mix of local ecosystems or land uses is repeated in similar form over a kilometers-wide area.'* According to the Swiss Federal Office for the Environment, landscapes result from the dynamic interplay of natural variables (e.g., bedrock, soil, water, air, climate, fauna and flora) with cultural, societal and economic factors. The most used definition is probably the one that resulted from the European Landscape Convention signed in Florence, Italy, on October 20th, 2020: *'Landscape means an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors'* (Article 1a) (Arsic 2015). Aware of the importance of sustainable landscapes at several levels, the Council established the promotion of 'landscape protection, management, and planning' as the main aim of the Convention.

Landscape pattern is linked to biodiversity and other ecological values (Uuemaa et al. 2013); therefore, understanding how landscape characteristics affect ecological processes and species' distribution is central to ecology (Kupfer 2012). The discipline of Landscape Ecology is essentially based on the notion that environmental patterns greatly influence ecological processes (Turner 1989) and, thereby, tries to describe and quantify landscape characteristics (Turner 2005). This is essential to understand and predict ecosystem services' sustainability and resilience, the availability of food resources, habitat suitability for animal species, and for the overall environmental assessment of landscapes (Lausch et al. 2015).

Significant and quick changes in landscape patterns often disrupt ecosystem functioning, interfering with critical ecological processes and, thus, jeopardizing their

capacity to provide vital services for the population and maintain biodiversity and overall ecosystem health (With 1997; Millennium Ecosystem Assessment 2005). Landscape changes can have such an impact that habitat loss and fragmentation are often pointed out as major causes of biodiversity loss (Jaeger et al. 2011) and, therefore, regarded as a central issue in Conservation Biology for some time (Wilcove et al. 1986; Saunders et al. 1991; Wiens 1996).

Landscapes are shaped and altered by ecological and anthropogenic processes (Lausch et al. 2015), and substantial changes, regardless of the origin, affect and may disrupt the natural balance of ecosystems. Human activity is almost always brought into the narrative not only due to its magnitude but also because it can be highly impactful in a short period of time. Moser et al. (2002) stated: "*Human influence changes landscapes significantly, and this significantly impacts biodiversity*". There is also a pragmatic reason for focusing on human-caused changes, in the sense that those are the ones we can influence the most. While it is possible to prepare for natural disasters, it is only possible to design and plan extensively for anthropic landscape changes.

When considering landscape changes, one usually addresses the land cover or land use types. Although 'cover' refers to the type of occupation the site has, and the term 'use' adds information regarding the magnitude of the activity (e.g., agriculture as a cover may have an intensive or extensive use), both terms are often used interchangeably or complementarily as land use/land cover (LULC).

1.2. The Patch-Corridor-Matrix Model, Composition and Configuration

Empirically, it makes sense to assume landscapes consist of delimited patches of different LULC. The patch-corridor-matrix model (PCMM) created in the 1980s does exactly that by considering that three basic elements form a landscape: patches, corridors, and matrices (Forman and Godron 1986). Every point in the landscape belongs to a patch, a corridor, or a background matrix, regardless of which LULC it falls in (Forman 1995a).

The reality is more complex than that, since there are also gradients with variation over space, without distinct boundaries. Under the PCMM, a forest may be represented by a discrete patch (or matrix) overlooking the internal and multidimensional variations, while

the Gradient Model (Müller 1998; McGarigal and Cushman 2005) would seek other aspects (measurable variables) that could portray different characteristics, such as the gradual change in the assemblage or density of tree species. A remote sensing-based analysis may uncover variations within an apparently homogeneous patch, indicating the spatial variability of potentially relevant features (e.g., physiological state, hydric stress, etc.). Although some landscapes may be better described by such a gradient-based concept (McGarigal et al. 2009), the majority is still well-suited to the PCMM framework, which despite its simplicity has many benefits and remains the dominant paradigm (With 2019). In human-dominated landscapes, the LULC classes tend to have clear boundaries (Lausch et al. 2015), making them especially fit for the PCMM. Regarding map production, multifunctional land use systems, such as the Portuguese *montado* and the Spanish *dehesa*, can constitute exceptions. They are human-shaped and, thus, require active management to endure, but their forest component is characterized by fuzzy boundaries and varying densities (Van Doorn and Pinto-Correia 2007; Godinho et al. 2016b). Within the PCMM framework, landscape structures can be described by quantitatively assessing their spatial patterns, which is valuable for the quantification of landscape functions (Bolliger and Kienast 2010) and for the quantification of ecosystem services (Syrbe and Walz 2012), among other uses.

To assess and interpret landscapes as discrete patches of different LULC classes (Forman 1995a), there are two main types of changes to consider: composition and configuration changes. Composition relates to what constitutes the landscape and its amounts (e.g., land cover types and how much) – What exists in the landscape. Configuration has to do with the shape and location of the elements – How it is distributed (McGarigal and Marks 1995). In other words, the analysis must consider both quantitative and geometric variations.

1.3. Landscape metrics and the concept of scale

Landscape patterns influence ecological processes (pattern-process relationship; Turner 1990; McGarigal et al. 2012), and numerous metrics have been developed to assess the composition and configuration of landscapes (Gustafson 1998, 2019). Moreover, the ecological significance of such metrics is a key topic in Landscape Ecology (Turner and

Gardner 1991). These tools enable us to compare different landscapes, quantify how they change over time, and investigate their relationships with other ecological processes (Uuemaa et al. 2009).

The thoroughness of the analysis and the complexity of some landscapes may dictate the need to use several metrics to capture the different qualities of spatial patterns (Tischendorf 2001). In fact, any given metric can only partially describe (one or two aspects of) the spatial patterns but cannot achieve a full description of the reality (Li et al. 2005). However, using many metrics increases the risk of correlation between them, which in turn may negatively affect the interpretation of the results (Li and Wu 2004). Given the large number of metrics available, it is a serious challenge for analysts to determine which metrics should be used to understand the different components of landscape structure. In order to prevent redundancy, it is advantageous to select the smallest number of (relatively) independent metrics that contribute sufficiently to the landscape structure quantification and description (Hargis et al. 1998; Cushman et al. 2008). Since there are no perfect metrics, drawbacks should be expected, acknowledged, and minimized as much as possible. For instance, scale dependence (Gustafson 1998) or the fact that different aspects of the landscape structure are correlated (Cushman et al. 2008) could be hard to overcome. Not being fully aware of what the metrics are actually measuring and how they respond to variations in landscape patterns (Hargis et al. 1998) is a well-known issue that can only be tackled with more theoretical and empirical understanding. Nevertheless, landscape metrics have many advantages, such as their simplicity and versatility, which allow rapid calculations of useful indicators, and the testing or simulation of LULC scenarios derived from environmental policies (Uuemaa et al. 2013).

Many metrics have been developed to assess the spatial arrangement of habitat patches, classes of patches and landscape characteristics like dominance, diversity, contagion, and fractal dimension (Haines-Young and Chopping 1996), but few were specifically suited to measure fragmentation. Exceptions to that are the metrics introduced by Jaeger (2000) and the 'proportion of landscape displacement from configuration' developed by Long et al. (2010). Jaeger (2000) characterises landscape fragmentation in a geometric perspective, using concepts such as the 'degree of landscape division', 'splitting index', and 'effective mesh size'. Long et al. (2010) quantify the relative contributions of

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forest loss and configurational change (fragmentation or aggregation) to the overall impact.

The concept of scale is one of the most used yet probably less understood, as it is prone to confusion. Scale is often used to refer to absolute size, relative size, resolution, granularity, and detail (Montello and Golledge 1998). For instance, the scale of a natural phenomenon has to do with the dimension it reaches; the analysis scale refers to the size of the analytical unit (AU) being used by the researcher; and the cartographic scale is the ratio between the dimension in a map and the actual dimension (Montello 1998). In cartographic language, the terms 'small scale' and 'large scale' refer to space reductions. 'Small scale' reflects a big reduction (e.g., 1:1.000.000), while 'large scale' means a small reduction (e.g., 1:2.000). In Landscape Ecology, scale can also concern the spatial and temporal dimension in which an organism, a pattern, or a process is identifiable (Farina 1998).

Taking the LULC as the study object, the methodological approach is often a function of the scale we intend to work with, depending on the concrete study goals. For instance, one or more phenomena may be analysed by considering a patch, all the patches of that LULC, or all the landscape patches in the study area. The analysis can be conducted at the patch level, the class level, or the landscape level, which entails different spatial ranges and has implications on the detail of the analysis. The case studies in this thesis vary in study area size (some consider countries, such as Portugal, Slovenia, or Germany, one considers a particular protected area) and in analytical scale (parishes and squares with different sizes).

1.4. Spatial processes in land transformation

Regarding its amount, a LULC category can remain unchanged, increase, or decrease. The configuration is more complex to assess, as the spatial arrangements of a certain amount of a LULC are virtually unlimited. However, whether through observation in real life or via simulation, it is possible to narrow the possibilities to a limited and workable number of spatial patterns.

Forman (1995a) identified five major spatial processes in land transformation (Figure 1a):

1. Perforation: the process of making holes in an object such as a habitat or land type;
2. Dissection: the carving up or subdividing of an area using equal-width lines (roads or other linear infrastructures);
3. Fragmentation: the breaking up of a large habitat or land areas into smaller parcels;
4. Shrinkage: the decrease in size of objects, such as patches;
5. Attrition: the disappearance of objects such as patches and corridors.
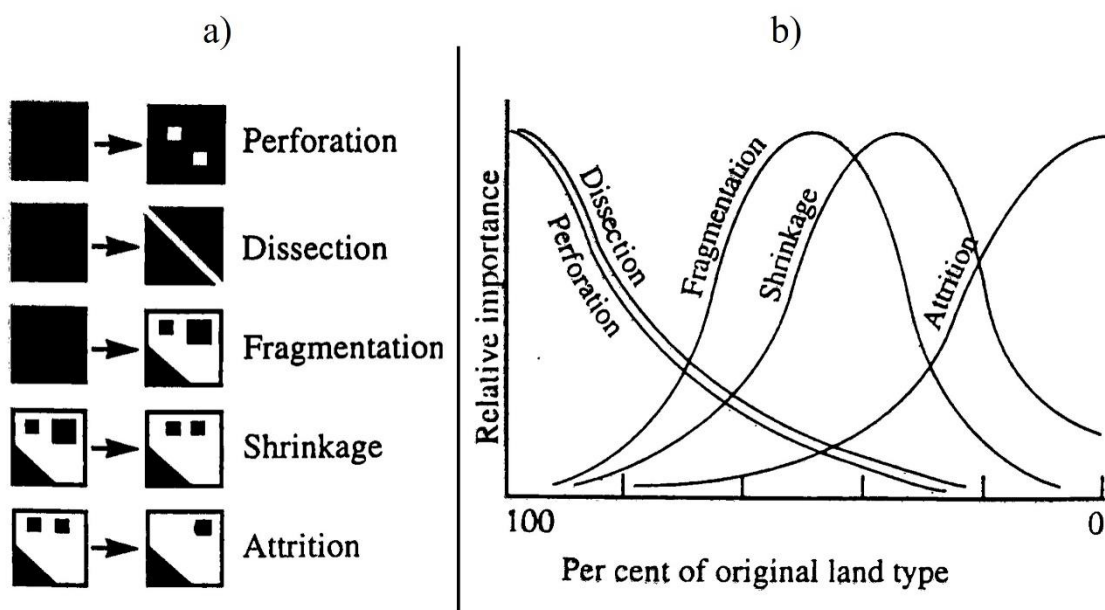
Figure 1 – Spatial processes of land transformation (a) list and graphical representation; (b) general spatial-process model indicating common phases of maxima and overlaps. (Adapted after Forman 1995a)

These processes are presented in the sequence in which they are likely to occur, given a habitat loss dynamic (Figure 1b). Starting with a delimited area or an AU totally covered by forest (which in this context is equivalent to the habitat), the most common way of beginning land transformation is to open clearings. The most common alternative is dissection. While perforation can happen by human hand (e.g., logging activity) or due to natural causes (e.g., storms or blowdown clearings), dissection is exclusively anthropic as it involves the removal of a linear strip of forest and conversion to other LULC (roads, railroads, irrigation channels, etc.). Next comes fragmentation, which conceptually does not differ much from dissection but produces a more uneven separation of the resulting pieces. In other words, instead of a clear, linear cut, it creates more complex patterns. In this line of thought, dissection can be considered a particular case of fragmentation.

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Perforation and dissection are more important in the early stages of habitat loss, but once the habitat is already perforated or dissected, further losses have a higher probability of producing fragmentation and shrinkage, which become the most relevant processes. If the amount of habitat keeps decreasing and approaches zero, patches disappear, making attrition the most relevant process in the final stage. Note that, although we can place these processes in a logical sequence, they overlap through the period of land transformation.

Based on Forman's work, Jaeger (2000) developed a similar approach. The first difference is at the conceptual framework level (hierarchical semantics), by using the expression 'phases of fragmentation process' instead of 'spatial processes of land transformation'. 'Fragmentation' is then used as a more comprehensive notion that comprises different phases and not as a single land transformation process. Other differences lie at the process level, where the spatial process originally designated 'fragmentation' became a phase called 'dissipation' and it was made clear it represents a combination of the phases 'dissection' and 'shrinkage'. There is a total of six fragmentation phases, as seen in Figure 2.
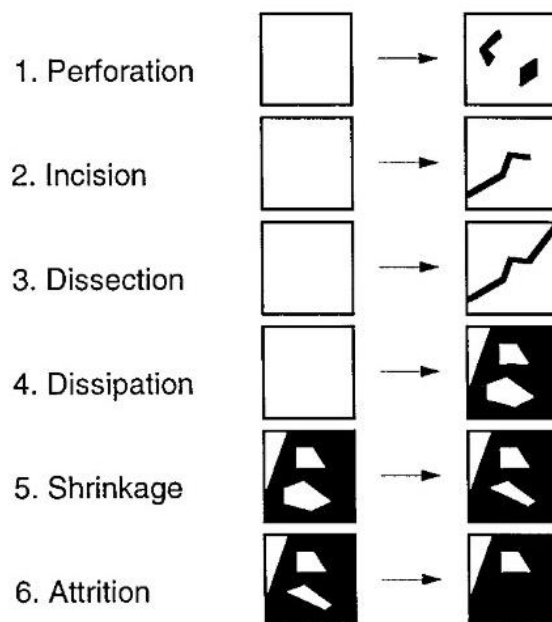
Figure 2 – Fragmentation phases (Jaeger 2000).

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Phase 2, 'Incision', is similar to phase 3, 'Dissection', as the original habitat suffers a linear 'cut', but in this case, the cut is not long enough to provoke a subdivision. The author's statement that "*From a purely geometric point of view, the dissipation phase has no separate meaning relative to the phases dissection and shrinkage.*"' can be disputed, as it depends on how geometry is assessed. In regard to the number of patches (NP), for instance, dissection and dissipation increase the NP, while shrinkage leaves the number of existing patches unchanged. Regardless, the author points out that a phase like dissipation makes sense to accommodate situations when dissection and shrinkage take place simultaneously and cannot be regarded as separate processes. This pertinent observation can stimulate further thinking on how to face special cases involving other phases besides these ones. It is worth to highlight that the number of different phases detected in the same place depends on how big the place is (AU size). The smaller the territory analysed, the less probable it is to identify several phases. Nevertheless, more than one phase can be detected in the same analysed area, and this should be addressed properly. First, every combination of phases should be considered; there could simultaneously be perforation and dissection, or incision and attrition, and so on, not to mention more than two phases at the same time. Would it be possible, and would it make sense, to come up with a designation for each combination? It is probably more appropriate to rely on a broader nomenclature able to promptly accommodate all variations and reserve the use of more specific designations for when such detail (added information) is necessary. Secondly, it would have to go beyond the problem of habitat loss and embrace a wider vision that also considers habitat gain.

Bogaert et al. (2004), in a more comprehensive approach, defined ten processes responsible for pattern change (Figure 3), looking not only at aspects related to habitat loss but also considering changes related to habitat gain:

1. Aggregation: the action or process of collecting units or parts into a whole; to bring or gather together into a whole; to fill gaps or open space;

2. Attrition: the reduction or decrease in the number of patches; the disappearance of patches;

3. Creation: the formation of new patches, which results in an increase of the total number of patches; the act of creating new patches; patch genesis;

4. Deformation: the change of patch shape, without patch size change; patch disfigurement;

5. Dissection: the carving up or subdividing of an area or patch using equal-width lines; sectioning of an area or patch; area or patch (sub)division;

6. Enlargement: the increase of patch size; patch size expansion;

7. Fragmentation: the breaking up of an area into smaller parcels, resulting in unevenly separated patches; the breaking up of extensive landscape features into disjunct, isolated, or semi-isolated patches;

8. Perforation: the process of making holes in an area or patch; gap formation; interruption of land cover continuity by the formation of openings;

9. Shift: patch repositioning; patch translocation;

10. Shrinkage: the decrease or reduction in the size of patches, without 'attrition'; progressive reduction of the initial land cover patch, ideally maintaining its original shape.
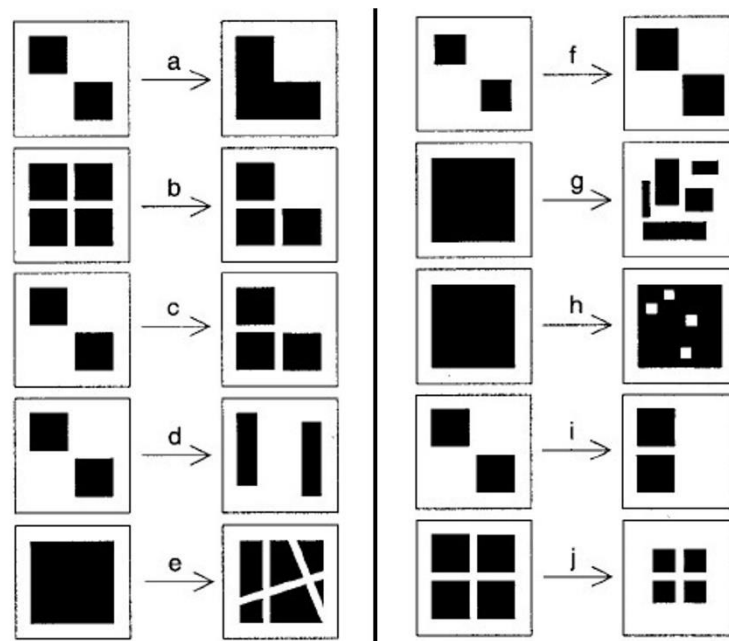
Figure 3 – Spatial processes in land transformation: (a) aggregation; (b) attrition; (c) creation; (d) deformation; (e) dissection; (f) enlargement; (g) fragmentation; (h) perforation; (i) shift and (j) shrinkage. (Modified after Bogaert et al. 2004)

In the list, besides the five processes by Forman (1995a) and shared by Jaeger (2000) despite minor semantic differences, there are three new processes equivalent but symmetric to others, which occur in a situation of habitat gain. 'Aggregation' is the opposite of 'fragmentation', 'enlargement' is the opposite of 'shrinkage', and 'creation' is the opposite of 'attrition'. Two more processes, strictly geometric (neutral towards habitat area), were included: 'deformation' and 'shift'. In comparison, the previous approaches contain spatial processes all involving habitat area change with a negative variation (habitat loss), while this proposal adds positive variation in habitat area (habitat gain) and the idea of configurational change without variation in the habitat area.

1.5. The importance of terminology

"Is a conceptually ambiguous and empirically multifaceted term fruitful as a generic description of human effects on landscapes?" (Haila 2002)

A semantic issue with relevant implications has been around in the Landscape Ecology sphere for some time (Farina 1998). It has even been pointed out that a new conceptual framework is needed because some terms and concepts in ecology have become vague due to incorrect use (Lindenmayer and Fischer 2007). Although assuming a part of the problem has to do with the loose application of metrics (Jaeger 2000), it seems that a major factor for confusion and misunderstandings lies in terminology. The vastly used expression 'habitat fragmentation' is one of, if not, the most problematic for two main reasons. First, often the study object may not be a habitat according to the proper ecological definition; second, there may be more than a single spatial process involved and yet it is usually called 'fragmentation'. Fahrig (2003) warned that looking at a single measure of change without separating the contribution of the composition from the contribution of the configuration makes studies difficult and sometimes impossible to compare. Although both types of changes co-occur, interact and even depend on each other (Lindenmayer and Fischer 2007), separating their independent effects is necessary to better understanding the impacts. According to Fahrig (2003, 2017), fragmentation is majorly understood as harmful for biodiversity because it is often assessed together with habitat loss, whose impacts are

stronger and mask the sometimes positive impacts of fragmentation. Related or not, actual habitat loss that always has negative impacts on the species under study and is considered by many to be the leading cause of decline in native species (Fahrig 1997; Foley et al. 2005) has received less research attention than expected (Fazey et al. 2005).

There will always be inconsistencies and room for progress, but a clear conceptual framework articulated with coherent terminology would ease the researchers' work, improve the communication among the community of experts and allow proper comparisons of methods and conclusions.

1.6. Nomenclature proposal

Despite the need for terminology clarification most researchers are aware of the importance of distinguishing between the effects caused by composition and configuration changes and have been conducting their studies accordingly. Some examples are the following: (i) the measure 'proportion of landscape displacement from configuration' that quantifies the relative contributions of forest loss and configurational change to the so-called forest fragmentation (Long et al. 2010); (ii) the comparison of separate landscape composition and configuration influence in aphid-parasitoid-hyperparasitoid interactions (Plecas et al. 2014); (iii) the study on the effects of habitat loss, fragmentation, and degradation on small mammals (Johnstone et al. 2014); (iv) the distinct contribution of landscape composition and configuration to catchment hydrological flows and variations (Liu et al. 2020); and (v) the structural connectivity of Protected Areas, which depends on their distribution besides other simple metrics such as the proportion of national territory classified (Ward et al. 2020). Nevertheless, it has been recently pointed out that the language associated with much of the habitat loss and fragmentation literature is still leading to erroneous conclusions (Fahrig 2017), with many authors still assuming in advance that the effects of fragmentation are generally negative.

Landscape dynamics are often equated to LULC changes and, for that reason, should be assessed considering both composition and configuration. A spatial process in land transformation, here also called Type of Dynamic (ToD), is a combination of the amount

and geometric change. Thus, its designation should be formed using the terms fragmentation, aggregation, gain and loss.

The problems are not limited to the ToD. The definition of the study object has also been inconsistent. The term 'habitat' has been used loosely to the point that it has become a vague concept (Mitchell and Powell 2003). 'Habitat' is a species-specific concept and therefore should be used in species-specific studies (St-Laurent et al. 2009). If the work is not focused on a particular species and consequently on the resources and conditions required for its presence (Hall et al. 1997), the term 'habitat' should be avoided. A common mistake is to consider 'habitat' the same as native vegetation (Fischer and Lindenmayer 2007) or immediately equivalent to a particular biotope or LULC category. Although the core definitions should remain clear, some variations could be accepted as long as they are explained. For instance, the Habitats Directive describes natural habitats as *"terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural"* (European Union 2013). Therefore, these areas are not necessarily species-specific, but delimited to encompass relevant faunistic and floristic values.

Ultimately, the landscape, land cover, ecosystem or otherwise object of interest to the study should be correctly identified to prevent the recurrent misuse of the term 'habitat'.

- Example 1: If the study focuses on an individual species and the study object is its habitat, the composition changes may reflect habitat gain or loss, while the configuration changes may reflect habitat fragmentation or aggregation.
- Example 2: If the study is about a forest and the change process occurring is fragmentation, the outcome is forest fragmentation, not habitat fragmentation.
- Example 3: If the study is about agricultural land area increment, the outcome is agricultural land gain, not habitat gain.

However, suppose that the type of forest or agricultural system constitutes the habitat of a certain species. In that case, it is correct to state that a habitat fragmentation occurred in the former case and habitat gain occurred in the latter.

The landscape dynamics designations are formed by the study object followed by the type of dynamic:

i) 'study object' + 'type of dynamic'

The type of dynamic consists of configuration change and composition change:

ii) 'type of dynamic' = 'configuration change' + 'composition change'

Thus, the resulting designation is formed according to the following structure:

iii) 'study object' + 'configuration change' + 'composition change'

Basic structure and examples:

'Study object' + 'type of dynamic'

= 'Study object' + 'configuration change' + 'composition change'

Example 1: Habitat + fragmentation + loss = Habitat fragmentation by loss

Example 2: Habitat + 0 + gain = habitat gain

Example 3: Agricultural land + 0 + loss = Agricultural lands loss

Example 4: Agricultural land + aggregation + gain = Agricultural lands aggregation by gain

Example 5: Forest + fragmentation + loss = Forest fragmentation by loss

Example 6: Urban area + 0 + gain = Urban area gain

Notes:

1. The broad nomenclature suggested here does not imply the abandonment of more accurate designations that add interpretative value. For instance, perforation is a particular case of loss, and it could make sense to use that designation to distinguish that unique pattern of loss in a given analytic context.

2. Because we are not used to regard aggregation and fragmentation as purely geometric phenomena, the ToD 'Aggregation by loss' and 'fragmentation by gain' can seem weird. It may be hard to associate fragmentation to habitat gain or aggregation to habitat loss. I suggest the alternative designations 'NP decrement by loss' and 'NP increment by gain'.

1.7. Goals and thesis outline

The main goal of this thesis is to build a landscape assessment tool that considers and distinguishes composition and configuration and whose outputs are spatial patterns formed by combinations of metrics. As secondary goals, contributing to the main one, there was the need to i) establish a framework and a precise nomenclature, ii) develop a method on top of which to build the tool and iii) demonstrate via concrete applications the tool's function and usefulness.

The goal structure of the thesis is as follows:

1. Conceptual framework and nomenclature
2. Method
3. Tool
4. Application

This first section (Chapter 1. '*Introduction*'), introduces the main theme of the thesis addressing specific topics and contains a nomenclature proposal for landscape dynamics, i.e., a system of names or terms and the rules to form them, which can be useful to promote a common terminology.

Chapter 2. '*Landscape Dynamic Typology – A method to assess land use / land cover changes*' presents an analytical method in line with the conceptualization and the nomenclature proposed in subchapter 1.6. It consists of a sequence of geoprocessing operations that lead to a landscape assessment based on spatial processes of land transformation.

Chapter 3. '*Tools to implement the Landscape Dynamics Typology*' introduces software tools to facilitate and automate the method application. A toolbox for ArcGIS, LDTtool, is introduced in subchapter 3.1., and a QGIS equivalent, LDT4QGIS, is presented in the subchapter 3.2. Both subchapters include descriptions of the tools and illustrative examples on how to operate them.

Chapter 4. '*Applications*' is dedicated to real case studies that used the method or the tools. It is divided in four subchapters: 4.1. concerns the management of invasive species in Portugal; 4.2. is about a study on grasslands dynamics in Slovenia; 4.3. involves the

montado land use system in Portugal; and 4.4. reports on the use of these tools by other researchers elsewhere.

Chapter 5. '*Conclusions*' is the final main section and summarizes the major conclusions stemming from the previous chapters, with a special focus on methodological and tool-related topics. Potential use cases for the tools are highlighted, known limitations are presented and future work is unfolded. The published papers directly related to this thesis can be found in the appendices.

CHAPTER 2. Landscape Dynamics Typology: A method to assess land use / land cover changes

This chapter is based on the article published in *Landscape Research* as:

Authors' contributions:

Rui Machado: Conceptualization, Analysis, Writing the original draft; Sérgio Godinho: Review; Janez Pirnat, Nuno Neves, Pedro Santos: Review and Supervising.

2.1. Background

Landscape transformations have been a central topic in Landscape Ecology due to their influence on ecosystem services and biodiversity conservation. Landscape dynamics can be very complex and difficult to analyse. A particularly difficult issue to assess is the contribution and the effects of landscape composition and configuration changes to the overall impact.

This work was inspired by the paper of Bogaert et al. (2004) as the goals are similar and the methodological approach follows theirs. By combining their foundations and the conceptual framework and nomenclature presented in subchapter 1.6, we obtained an improved method for landscape dynamics assessment. The main goal was to propose a method for assessing habitat composition and configuration changes considering and identifying the processes of habitat loss/gain and habitat fragmentation/aggregation.

2.2. Materials and Methods

2.2.1. Landscape Dynamics Typology establishment

Landscape Dynamics Typology is a classification system in which landscape changes are aggregated according to the processes that originate them. It uses binary landscapes, imposing a simplification in which a landscape is composed by two classes only: the LULC under study and other class that encompasses the remaining LULC. The dynamics are obtained by considering how changes in composition and configuration are depicted by certain metrics. Following the advice by Cushman et al. (2008), of using a smallest number of appropriate metrics to quantify landscape structure, we used the LULC class area to measure composition and the number of patches (NP) to assess configuration. Class area is a measure of landscape composition which reveals how much of the landscape is comprised by a particular patch type and NP represents a measure of the subdivision of the patch type and is linked with a number of ecological processes (Cumming and Vernier 2002; McGarigal et al. 2012).

Some dynamics are relatively easy to understand:

> i) If the amount remains the same and the NP increases, we are facing a fragmentation *per se* event (pure geometric variation).

> ii) If the amount increases and NP decreases, it reflects an aggregation of patches due to an amount gain (geometric variation caused by an amount variation).

However, there are other patterns that reflect different metric combinations that are conceptually difficult to interpret. To answer simple questions that arise after the analysis, such as 'How far can this trend go?' or 'How will the landscape look like if the agro-forestry areas increase?' it is necessary to develop a protocol/tool able to perform simulations.

Using sampling units such as squares, we assume, for convenience, that the squares with amount gain will continue gaining, leading to a 'total cover' scenario, and the squares with amount loss will keep losing, eventually reaching a 'no cover' situation. In each of these two paths towards the extremes, the squares can pass through different dynamics. Finally, a dichotomous key was also created to assist the user to easily and promptly classify a dynamic.

2.2.2. Case study, land cover data and procedures

We conducted a spatiotemporal analysis focused on the dynamics of the agro-forestry areas in the Portuguese mainland territory between the years 1990 and 2006. Agro-forestry areas were obtained using ArcGIS 10.2 (ESRI 2013) software and three shapefiles available online: two CORINE land cover shapefiles (years 1990 and 2006) (Caetano et al. 2009) and the Portuguese Administrative Boundaries Official Map - 2014 version. According to the CORINE land cover nomenclature, the agro-forestry areas encompass annual crops or grazing land under the wooded cover of forestry species. This makes this land cover category not completely equivalent, but greatly coincident to one of the most important ecosystems in the Iberian Peninsula, the *montado*, in Portugal and *dehesa*, in Spain. The *montado* has suffered a decrease in the last decades (Godinho et al. 2016c), which make it suitable to demonstrate the Landscape Dynamics Typology (LDT) method.

Aiming to identify dynamics not only globally (at the landscape scale) but also at a more detailed scale, a sampling scheme was needed. For that, we built a 10 km x 10 km grid and used each square as AU, where area and NP of agro-forestry category were calculated. The task involved the following steps: building a 10 km x 10 km grid; extracting the agro-forestry areas from both CORINE land cover shapefiles; calculating agro-forestry area and NP, for the two dates, in each square; joining the results of both calculations into a single table; calculating the variation between years for each metric in each square; and assigning each square to a dynamic (Figure 4). Selection and calculation tools (e.g., 'select by attributes' and 'field calculator') were used to automate the process avoiding the need to replicate the steps for each square. Note that for conceptual reasons and for better understanding we mentioned binary landscapes but operationally only the LULC of interest is analysed while the other category is discarded or ignored.
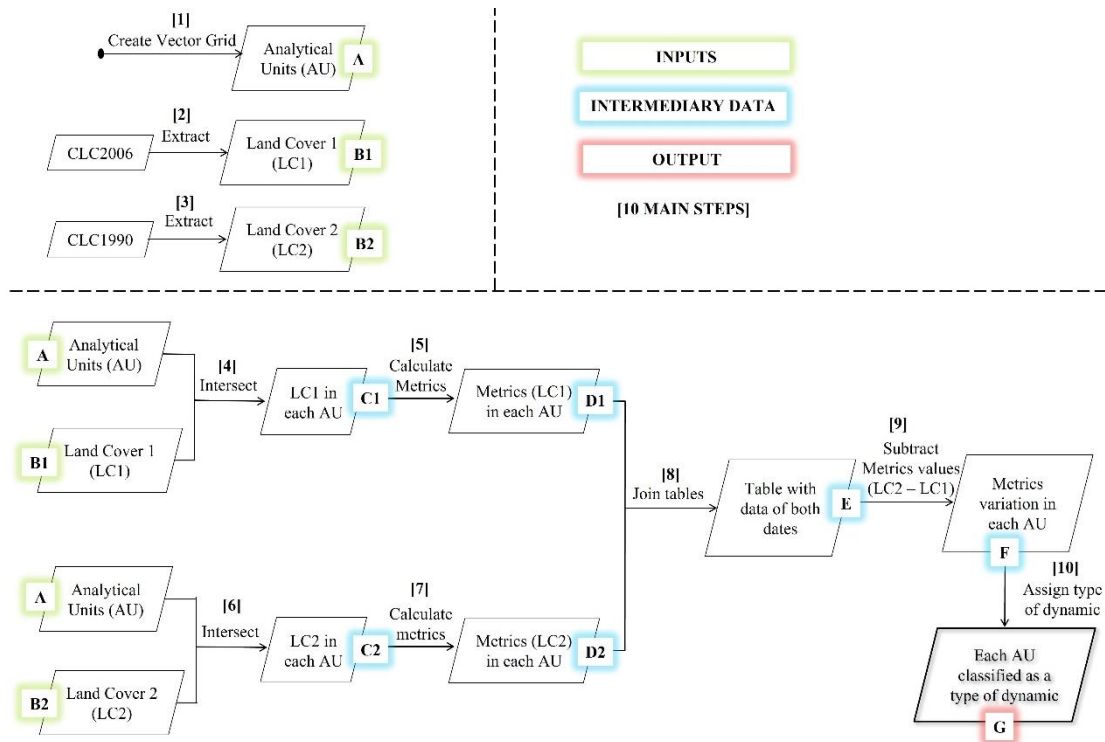
Figure 4 – LDT implementation steps.

2.3. Results and Discussion
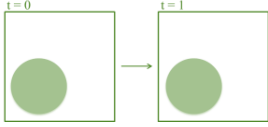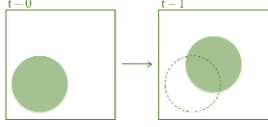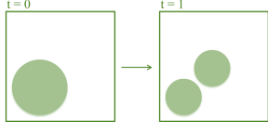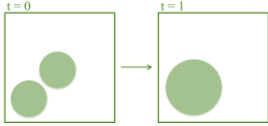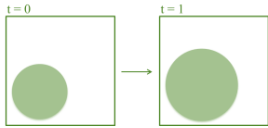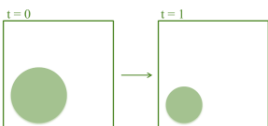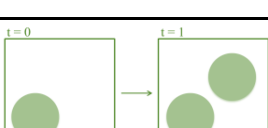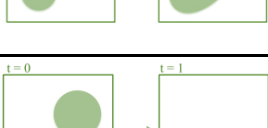
There are two major categories of results: those concerning the LDT and those regarding the practical implementation.

2.3.1. Landscape Dynamics Typology

The LDT is a list of possible landscape changes, defined by the metrics' behaviour. Besides the metrics' combinations, it contains the ToD, designations, and spatial pattern representation of the change (Table 1). These elements were also integrated into a dichotomous key useful for a quick analysis of the transformation processes acting in the landscape (Appendix 1).

Table 1 - Landscape Dynamic Types. ΔA – area variation; ΔNP – number of patches variation. (A) amount related dynamics; (G) geometric related dynamics.

If	and	Type of Dynamic	Spatial representation	
ΔA=0 ^ Symmetrical difference = 0	ΔNP=0	A - No change		
ΔA=0 ^ Symmetrical difference > 0	ΔNP=0	A1 - Spatial shift		
ΔA=0	ΔNP>0	B - Fragmentation *per se*		(G)
ΔA=0	ΔNP<0	C - Aggregation *per se*		(G)
ΔA>0	ΔNP=0	D - Gain		(A)
ΔA<0 ^ Symmetrical difference output **is not** completely contained in the original patch(es)	ΔNP=0	E - Loss		(A)
ΔA<0 ^ Symmetrical difference output **is** completely contained in the original patch(es)	ΔNP=0	E1 - Perforation		(A)
ΔA>0	ΔNP>0	F - NP increment by gain		(A,G)
ΔA>0	ΔNP<0	G - Aggregation by gain (NP decrement by gain)		(A,G)
ΔA<0	ΔNP<0	H - NP decrement by loss		(A,G)
ΔA<0	ΔNP>0	I - Fragmentation by loss (NP increment by loss)		(A,G)

Besides the ToD A, that symbolises a situation where no changes occurred in the LULC of interest, the dynamics can be divided in three classes: i) geometric related, ii) amount related and iii) both geometric and amount related. The ToD A1 – Spatial shift represents a patch translocation (same area, same patches but in a different place). The ToD B (fragmentation *per se*) and C (aggregation *per se*) represent pure geometric changes. These are not easy to find in nature because they are supposed to occur without any amount change. Nevertheless, we kept them in the LDT because they have theoretical justification as a landscape could be fragmented without amount change if the patches could be physically rearranged (Wang and Cumming 2009). A possible use case for such ToDs is scenario building for environmental compensation or ecosystem/habitat impact mitigation or enhancement. ToD D (gain) and E (loss) regard pure amount changes. These are expected to occur frequently and constitute the basis for the dynamics that display both composition and configuration dynamics. ToD E (Perforation) is a particular case of Loss in which the lost amount originates a clearing. ToD F (NP increment by gain), G (aggregation by gain), H (NP decrement by loss) and I (fragmentation by loss) reflect geometric changes prompted by amount variations.

The LDT forecast scheme shows how the dynamics are interconnected in a trajectory of amount gain or loss (Figure 5). In the presence of area gain: ToD G (aggregation by gain) will continue eventually until it reaches a 'total cover' situation; ToD F (NP increment by gain) can progress to a point when the amount is such that it will trigger the ToD G (aggregation by gain); ToD D (gain) has two possible trajectories before it reaches total cover: if there is only one patch, it can continue growing to the maximum extent; if there are multiple patches, their expansion will at some point create aggregation (ToD G) and after that evolve to total cover. In the presence of area loss: ToD H (NP decrement by loss) is going directly to a 'no cover' scenario when the last patch disappears; ToD I (fragmentation by loss) is a predecessor of the ToD H, meaning that the fragmentation produces small patches that will disappear thereafter; ToD E (loss) can go directly towards a 'no cover' situation if there is a single patch in the area, or towards the ToD H before that, if there are multiple patches. Finally, aggregation *per se* (ToD C) and fragmentation *per se* (ToD B) are not mentioned in the LDT forecast scheme because both processes only occur when ΔA=0.
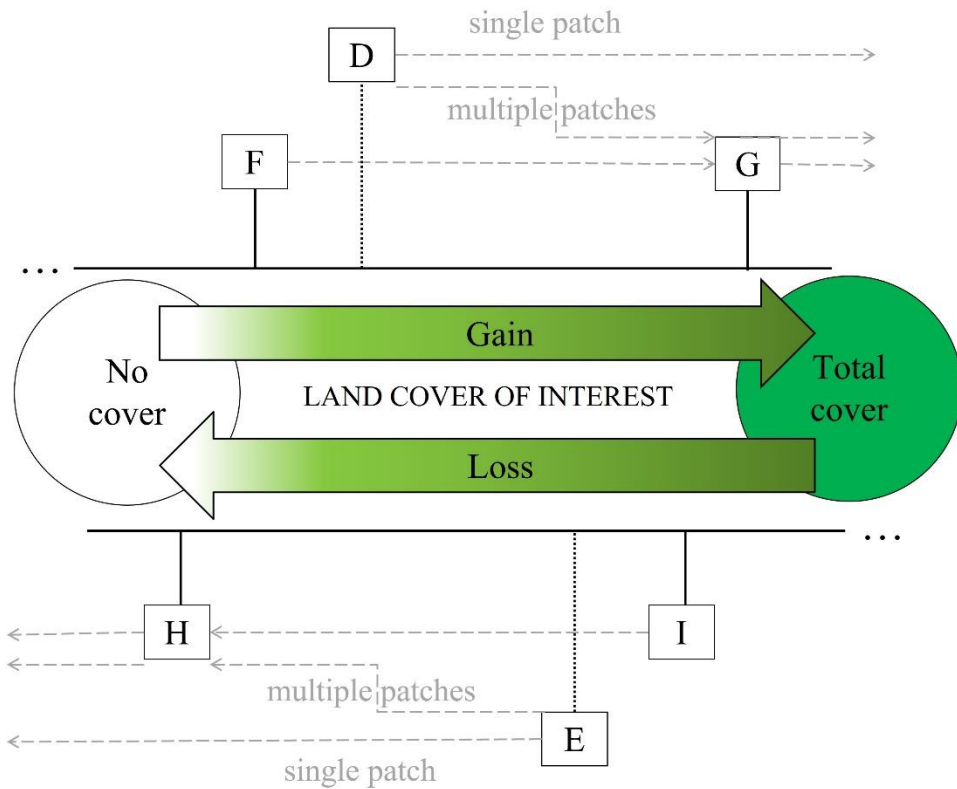
Figure 5 – LDT forecast scheme. D - gain; E - loss; F - number of patches incremented by gain; G - aggregation by gain; H - number of patches decrement by loss; I - fragmentation by loss.

LDT is not a sophisticated scenario simulation tool and cannot thoroughly predict what the metrics values will be on a given date. Instead, the forecast scheme is useful to preview a trajectory of a LULC class, assuming the trend will continue and ignoring external drivers of and obstacles to land transformation, like what neutral landscape models do. Neutral landscape models (also called null models or random models) generate random habitat patterns, not accounting for drivers that might affect landscape pattern. These are useful to predict how a landscape would look like if no processes affected the distribution of a particular LULC (Donovan and Strong 1997) and provide a reference point against other considered alternatives (Gotelli and Graves 1996). Forman (1995) organized the spatial processes of land transformation in a temporal axis because some processes precede others (Figure 1b). The forecast scheme is the equivalent in the LDT framework.

2.3.1.1. Limitations

Although the dynamics considered here are realistic, such an exercise is necessarily a simplification of reality because landscape dynamics can be very complex and thus hard to fully replicate with this or any other approach. The LDT can provide valuable outputs and information but has limitations (e.g., uses binary landscapes, oversimplifying reality) and should only be implemented within its range of applicability.

2.3.1.1.1. Modifiable Areal Unit Problem

Despite the advantages provided by the use of AU, we must highlight the modifiable areal unit problem (MAUP) (Gehlke and Biehl 1934; Openshaw 1984). The MAUP is a long-standing issue in geography and spatial analysis related to the fact that the areal units can be set arbitrarily. Thus, results based on aggregated data following areal units can change if aggregated under different areal units (Waller and Gotway 2004). It has no optimal solution and is usually addressed according to the study context. Openshaw (1984) proposed that areal units should match the optimal spatial variance or maximize a given statistic. Swift et al. (2008) reported that areal units generated from Voronoi tessellations (Thiessen polygons) can be effective to reduce aggregation bias. Butkiewicz et al. (2008, 2010) introduced a geospatial visualization method to help mitigate the MAUP effects and (Long et al. 2010) suggested a multiscale approach to minimize scale-related bias.

In our example, we used squares as AU to study agro-forestry areas. If the squares had a different size and/or a different location, the content in terms of agro-forestry areas would also be different (scale effect and zonation effect, respectively). In an actual study, rather than an illustrative example like this, this issue would have to be addressed whether by trying to minimize the differences or testing the results to find out if the discrepancy is acceptable to carry out the analysis. Examples and further discussion about MAUP can be found in '3.1.3.2.1. Square size selection – Tackling MAUP' section.

2.3.1.2. Strengths and implementation scope

LDT's more visible strength is that it is based on ordinary tools and concepts like landscape metrics and transformation processes well established in the literature. Concerning its applicability, LDT is versatile in the sense that it systemises a procedure but does not imply the use of specific data formats or software. Besides that, the steps are editable and the parameters adjustable. For instance, when using squares, they can be designed in terms of size or location to fit the user's purpose. Finally, it is important to stress how the results are easy to interpret because it all rests on metric values, their variations, and even more pragmatic, actual spatial patterns (ToD). Moreover, geographic information software allows the visualization and representation of the information while preserving its spatial attributes.

The LDT was designed to detect landscape transformation processes and thus its range of application is vast. The two most evident applications for LDT are monitoring tasks in a diagnosis context or forecasting in a territory planning framework. Whenever the LDT is well adapted to a certain study subject and context (e.g., a certain land cover class from a specific region, obtained from a given cartography, etc.) it can be chosen as a monitoring tool for that territory. Depending on the scope and depth of the analysis outline, LDT can either provide the final information required or, can be the first step to evaluate more detailed processes related to biodiversity, resource planning, ecosystem services, among others. The principles and methods of Landscape Ecology have been used by landscape planners and architects to preserve, restore and enhance biological diversity (Collinge 1996) and LDT can help in that mission.

2.3.2. Case study

From the total 1.012 squares, agro-forestry areas in 1990 or 2006 were found in 417. They were present in 416 squares in 1990 and in 413 squares in 2006. Agro-forestry areas were present in both 1990 and 2006 in 412 squares. In the 1990 map, there were 2977 patches of agro-forestry systems with a total area of 634.862 ha and in the 2006 map these values were 3028 patches and 621.495 ha. The global balance shows a variation of +51

patches and -13.358 ha, suggesting the occurrence of fragmentation due to a loss of agro-forestry amount at the study area scale.

The use of AU provides more detailed data such as the number of squares with area loss, gain or maintenance (Figure 6a) and the number of squares with NP loss, gain or maintenance (Figure 6b). The number of squares displaying negative area variation was much larger than that displaying positive area variation. Almost half the squares showed an agro-forestry areas loss between 1990 and 2006. Regarding the NP variation, most squares remained unaltered. Besides those, more squares revealed NP increment than decrement, which suggests more fragmentation than aggregation.

Regarding agro-forestry areas variations among squares, an analysis based on area size classes provided new insights about the studied land cover dynamics (Figure 6c). The number of squares with loss is much larger than those with gain (200 against 104), which suggest the loss was not confined to a certain region but instead was spread throughout the study area. An additional perspective is provided by Figure 6d), which shows the number of squares assigned to each dynamic. More than a quarter of the 417 squares with presence of agro-forestry systems showed no changes between the two dates (ToD A). Pure geometric variations ('aggregation *per se*' - ToD B and 'fragmentation *per se*' - ToD C) were not found at all. There were more squares classified as dynamics related to loss (NP decrement by loss – ToD G, Loss – ToD H and Fragmentation by loss – ToD I) than related to gain (NP increment by gain – ToD D, Gain – ToD E and Aggregation by gain – ToD F). The implementation of the LDT using AU also revealed where each dynamic was acting in the study area (Figure 7).
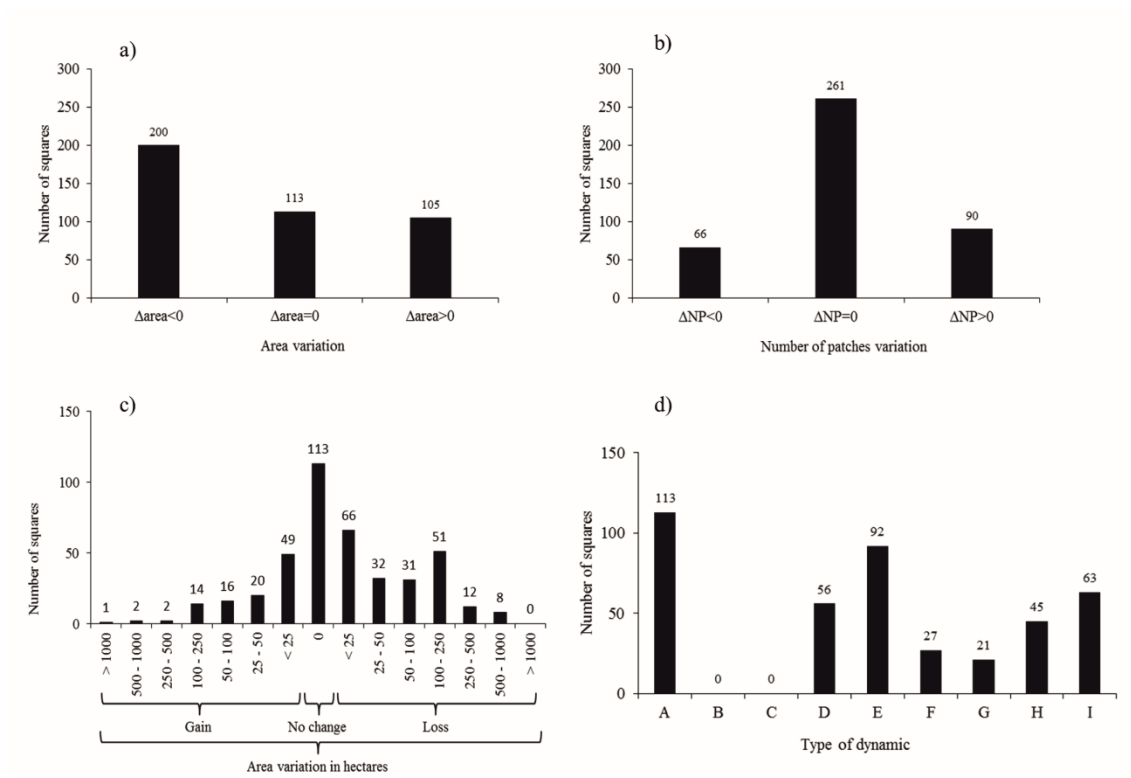
Figure 6 – a) Number of squares vs. area variation, (b) number of squares vs. number of patches variation; c) number of squares vs. area variation in classes and (d) number of squares vs. assigned type of dynamic.
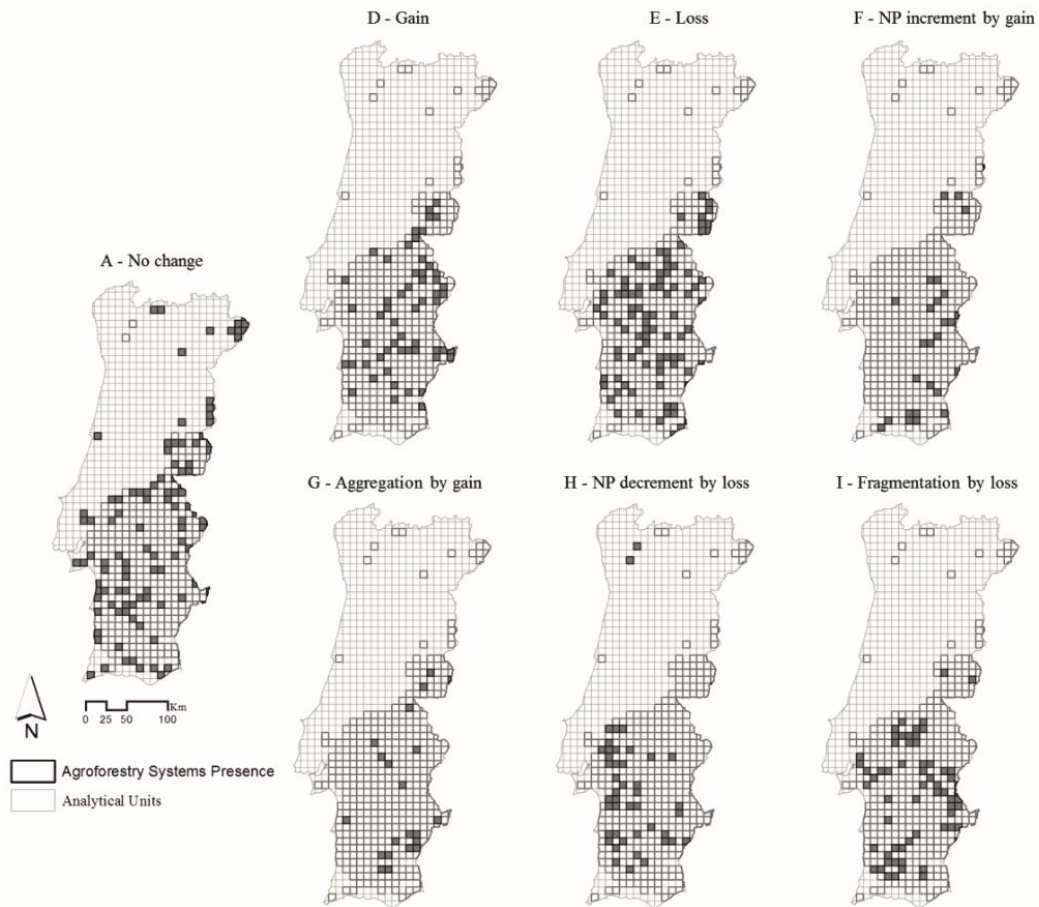
Figure 7 – Spatial distribution of different types of agro-forestry systems dynamics identified between 1990 and 2006.

From 1990 to 2006, both an amount decrement and a NP increment of agro-forestry areas occurred, at the study area scale, suggesting the landscape transformation was characterised as fragmentation due to area loss. This conclusion reflects an oversimplification of the transformations present in the territory because it hides several dynamics masked by the dominant one. When we look at study area global values, the expectable transformation processes are fragmentation by loss, NP decrement by amount loss, aggregation by amount gain or NP increment by amount gain. This leaves out the raw gain and loss, which are only traceable at larger scales as they usually concern single patches. If a patch expands or compresses, but not enough to merge it to or split it from another patch, this is a composition related process without relevance in the configuration. A gain, if big enough, can lead to an aggregation and a loss, if big enough, can lead to

fragmentation. To sum up, the use of a single value for an entire landscape or study area is not viable to track the pure gains or losses, because of its aggregate effect.

In this example we used maps from two dates only. The observed variations between 1990 and 2006 reveal a loss of agro-forestry areas. However, an intermediate map could show us that the agro-forestry areas were even less on that date and in 2006 were recovering. The values would be right, but the trend would not. Searching other works to verify our results and interpretations we confirmed that the agro-forestry areas decrement in the Iberian Peninsula in the last decades (Plieninger 2006; Godinho et al. 2016c) is a well-known phenomenon.

It is possible to use AU such as squares, hexagons or even irregular forms to identity dynamics at a local level. The form and size of the AU can be defined by operational criteria (e.g., other data or variables are available in a defined format and a consistent approach is needed), ecological criteria (e.g., the unit must be big enough to accommodate the phenomena under study), administrative-related (e.g., relationship to land management planning units or ownership.), etc. This can be understood as a type of spatial generalization with the purpose of better representing the data for a specific analytical purpose. Data is often collapsed and aggregated to make it more workable, to gain understanding of the studied phenomenon and to uncover patterns masked by the noise usually found in observations. Although the use of AU is neither risk-free nor always recommended, it is frequently used because it can be useful to some extent (e.g. Cumming and Vernier 2002).

The analysis based on squares reinforced what was revealed by the overall values for the entire study area, a trend of agro-forestry amount loss and a NP increment. Based on the raw values for the whole landscape we could only speculate this combination would result in fragmentation due to loss. Using squares, we can achieve more detailed results because we can identify not only fragmentation by loss (ToD I) but also NP decrement by loss (ToD H) and pure loss (ToD E), not considering the squares with gain. There were 92 squares with loss, 45 with NP decrement by loss and 63 with fragmentation by loss (Figure 6d). NP decrement by loss and fragmentation by loss are more likely to occur in the presence of greater losses. The larger number of squares with loss (ToD E) when compared with those of ToDs H and I indicate that in most squares the losses were small. It is more likely for small landscape changes to take place than big ones, thus it is expected to find fewer squares showing big changes and more squares showing small changes. This pattern

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is identifiable, although with some exceptions, in the different size classes represented in Figure 6c).

Long et al. (2010) were able to demonstrate that their metric Py, can differentiate between landscapes primarily experiencing forest loss, concurrent forest loss and configurational change, and those primarily experiencing configurational change. In their study significant forest loss occurred more frequently than configurational change. Our results are similar in the sense that we detected pure amount changes, a variety of configuration changes originated by amount changes but no strictly geometric transformations.

The case study is demonstrative and does not involve an actual deep analysis of the agro-forestry areas or the drivers that are influencing them. A thorough study could use the information provided by the LDT and consider the ToD locations and probable evolutions, study them, relate them with other variables of interest and come up with valuable insights and practical recommendations for management.

2.4. Conclusions

We enhanced an existing approach designed to assess landscape transformation via metrics combination. This new method produces detailed results concerning the location of the different dynamics in a landscape, providing actionable knowledge relevant in topics such as biodiversity preservation, invasive species management, and overall LULC related planning. The main achievements of this work are:

- The establishment of a 'Landscape Dynamics Typology'. This is a list of possible types of dynamics (transformation patterns) that can take place in a landscape.

- The supply of a diagnostic tree, which is a valuable tool to easily find out which dynamics are taking place in a given area.

- A case study regarding agro-forestry systems in Portugal, illustrating how the LDT can be applied in landscape analysis.

Future improvements can include the adoption of new paradigms or fitting the LDT approach in broader analytical contexts. Through practical experiments it will be possible to evaluate the MAUP effects, to test operational specifications like the minimum mapping unit and to include other metrics.

Applying the method to the case study was a long and iterative process that involved testing several geoprocessing tools and feature settings. The case study was operationalized using ArcGIS's model builder and during the process it became clear that the transition of LDT as a protocol to an ArcGIS toolbox was needed to facilitate its usage and increase its reproducibility.

2.5. Landscape Dynamics Typology and other frameworks

In the 'Introduction' section, different approaches to spatial patterns and landscape dynamics were presented and briefly analysed. It was also explained why a common framework with a clear terminology could help prevent the misuse of concepts and contribute to harmonizing approaches and methods in order to obtain comparable outputs. LDT is a method that emerged from an iterative and interactive essay of idealizing a framework and a terminology. It is relevant to understand what LDT's place in the global picture is and how it relates to the existing paradigms. LDT is broad enough to accommodate Forman's five spatial patterns, Jaeger's six phases, and Bogaert et al. ten spatial processes (Table 2). For convenience, I will use the designation 'ToD' to refer to all of these. LDT is the only one considering strictly geometric changes like fragmentation *per se* and aggregation *per se*, which can only be done if there is an underlying intent to disentangle the composition and configuration aspects. The most similar approach is the one by Bogaert et al., since it also considers gain of habitat. Forman's and Jaeger's are focused on the specific problematic of habitat loss and do not include any ToD involving habitat gain. For this reason, some LDT's ToDs have no equivalent in these two approaches. LDT has 11 ToD but since 'Perforation' is a specific case of 'Loss', it is correct to assume that LDT is composed by ten main ToD, that can be reduced to nine if the user finds 'No change' pointless. 'Perforation' emerged, as an extra feature, while software was being developed, and not as a pure ToD derived from the basic algebraic operations based on the variation

of area and NP. More ToD can be added in the same way, to include other patterns or processes described by other authors. Being comprehensive may imply losing some accuracy, and that explains why LDT does not automatically include specific ToD such as 'Deformation', 'Incision' and 'Dissection' and instead classifies them as 'Spatial shift', 'Loss', and 'Fragmentation by loss', respectively.

Table 2 – Correspondence between Landscape Dynamic Typology's types of dynamics, Forman's spatial patterns of land transformation, Jaeger's phases of fragmentation process and Bogaerts et al. spatial processes in land transformation.

Machado et al, 2018 LDT	Forman, 1995	Jaeger, 2000	Bogaert et al, 2004
No change	-	-	-
Spatial Shift	-	-	Shift, Deformation[1]
Fragmentation *per se*	-	-	-
Aggregation *per se*	-	-	-
Gain	-	-	Enlargement
Loss	Shrinkage	Shrinkage, Incision[2]	Shrinkage
Perforation	Perforation	Perforation	Perforation
NP increment by gain	-	-	Creation
Aggregation by gain	-	-	Aggregation
NP decrement by loss	Attrition	Attrition	Attrition
Fragmentation by loss	Dissection[3], Fragmentation	Dissection[3], Dissipation	Dissection[3], Fragmentation

[1] Deformation is a particular case of 'Spatial Shift' in which the number of patches and area remain the same, but the patch shape changes.

[2] Incision is a particular case of 'Loss' in which the number of patches remain the same and area was lost in a linear form, like a corridor, but does not go further enough to subdivide the patch.

[3] Dissection is a particular case of 'Fragmentation by loss', in which area is lost in a linear form subdividing the original patch and increasing the number of patches.

CHAPTER 3. Tools to implement the Landscape Dynamics

Typology

This chapter introduces the developed software tools that apply the LDT principles to geospatial data and automate the method implementation. The subchapter 3.1. presents LDTtool, a toolbox for ArcGIS. The subchapter 3.2. concerns LDT4QGIS, its equivalent for QGIS (www.qgis.org).

3.1. LDTtool: a toolbox to assess landscape dynamics

This subchapter is based on the article published in *Environmental Modelling and Software* as:

Authors' contributions:

Rui Machado: Conceptualization, Software, Testing, Writing the original draft; Roy Bayot: Topic expertise (Python); Sérgio Godinho: Validation; Janez Pirnat, Pedro Santos, Nuno de Sousa-Neves: Review and Supervising.

3.1.1. Background

The focus on accounting for both amount and geometric changes in landscape assessment procedures is the foundation of the Landscape Dynamic Typology method (Machado et al. 2018). LDT, although simple to understand, requires a heavy workload to implement manually. To tackle that, we introduce the LDTtool, an ArcGIS toolbox designed to facilitate and automate the application of the LDT method.

Many metrics, methods and software with distinct specificities have been developed to measure landscape characteristics. Some examples of well-known software are FRAGSTATS (McGarigal et al. 2012), Conefor Sensinode 2.2 (Saura and Torné 2009), Patch Analyst (Rempel et al. 2012), Circuitscape (McRae et al. 2013), Graphab (Foltête et al. 2012) and Land-metrics DIY (Zaragozí et al. 2012). Despite the variety of landscape software available (Steiniger and Hay 2009; Steiniger and Hunter 2013), new ones keep being developed in order to improve the existing analytical capabilities (e.g. landscapemetrics by Hesselbarth et al. (2019); gDefrag by Mestre et al. (2019)). LDTtool was built exclusively to assist with the LDT implementation and does not aim to replace any of this software. We provide a comprehensive description of the toolbox as well as an illustrative case study concerning the olive grove dynamics that have been occurring in southern Portugal.

3.1.2. LDTtool description

The LDTtool is a python-based (www.python.org) add-on ArcGIS toolbox operational in ArcCatalog and ArcMap as well as in ArcGIS Pro. It was developed using the 10.6 version and updates to future versions will be assured to keep the toolbox usable. It comprises eleven scripts (Figure 8):

- The tool 1 – "Landscape Dynamic Types" is composed by eight scripts and runs the core LDT steps to calculate the ToD. Those whose name contains '2M' perform analysis between two analytical moments and those whose name contains '3M' do so for three analytical moments. The suffixes 'Squares' and 'Districts' indicate the AU will be automatically generated squares or user-provided boundaries. For each of these tools there is a similar one which only difference is that it searches for the ToD 'perforation' inside the AU. This is optional and not standard because when it is assigned to an AU, it means that it occurred but does not mean it was the only area loss that occurred, so extra caution is needed to avoid misinterpreting the results. This is not a problem when the same procedure is applied to the entire LULC class, using the tool for perforation calculation, mentioned below.

- The tool '2 – Forecast' calculates a hypothetical scenario assuming the ongoing trends will persist. The forecast tool considers how the ToD can evolve from one to the other (Machado et al. 2018).

- The tool '3 – Perforation' is an accessory tool to geoprocess, at the class scale, the spatial pattern 'Perforation'. It shows where perforation happened between two analytical moments.

- The tool '4 – Gained and lost patches' is an accessory tool to identify and locate the places where amounts of the LULC category of interest were gained and lost between two analytical moments, including new individual patches or individual patches that disappeared. It produces four output feature classes (FC): all gains, gained patches, all losses and lost patches.

Additionally, two pre-arranged symbology files are provided and can be loaded and applied to the output files.
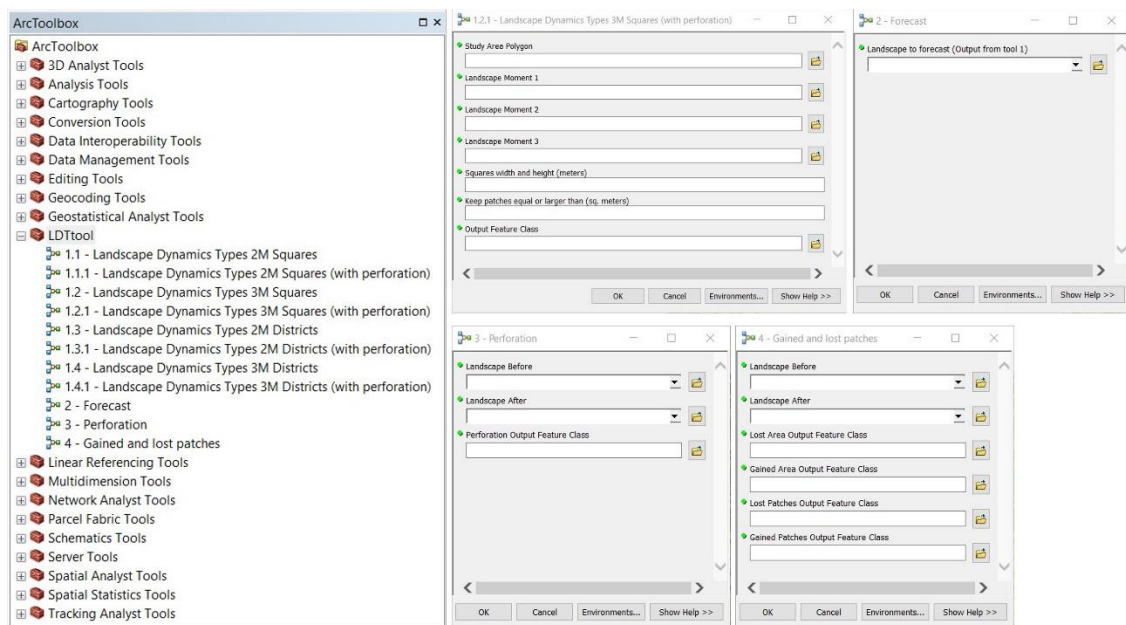
Figure 8 – LDTtool structure in ArcToolbox.

3.1.2.1. Preliminary steps, inputs and settings

Relevant recommendations to prevent errors or malfunctioning are: (i) to use the same Coordinate Reference System (CRS) in the data frame and all the input elements; (ii) to use FC (requires a geodatabase) instead of shapefiles and (iii) delete unnecessary attribute fields. The landscape FC must contain only one class with the polygons representing the object under study (habitat, biotope, LULC category, etc.). The inputs quality also affects the overall quality of the analysis. For that reason, spatial and temporal resolution should be adequate to assess the phenomena under study and represent accurately the landscape evolution.

The following preliminary steps are essential to ensure the toolbox correct functioning:

1. Create a File Geodatabase.
2. Select it as the default geodatabase.
3. Import the FC/shapefiles into the geodatabase. The FC should be the landscape moment 1, landscape moment 2 and landscape moment 3 (optional). Regarding the analytical units, the FC are districts or the study area boundary (for automatic square generation).
4. Add the LDTtool toolbox to ArcToolbox.
5. Select the Geodatabase as "Current workspace" and "Scratch workspace" in *Geoprocessing menu, Environments, Workspace*.
6. Select the Geodatabase as "Current workspace" and "Scratch workspace" in the ArcToolbox *Environments, Workspace.*
7. Confirm the paths are correct within each tool by Right-clicking; *Properties; Environments;* check the *Workspace* box*, Values* button*.*

The inputs and settings required to run the tools are the following:

Tool 1 – Landscape Dynamics Types

- *Study Area Polygon*: Polygonal FC containing the study area boundaries.
- *Districts:* Polygonal FC containing the districts' boundaries.
- *Landscape Moment 1*: Polygonal FC of the landscape in moment 1.

- *Landscape Moment 2*: Polygonal FC of the landscape in moment 2.

- *Landscape Moment 3*: Polygonal FC of the landscape in moment 3.

- *Squares width and height (meters):* Analytic square size.

- *Keep patches equal or larger than (square meters)*: Minimum patch size to be analysed.

- *Output Feature Class*: Name and path of the output file.

Tool 2 - Forecast

- *Landscape to forecast (Output of tool 1)*

Tool 3 – Perforation

- *Landscape Before*: Polygonal FC of the landscape in the earliest moment of analysis.

- *Landscape After*: Polygonal FC of the landscape in the latest moment of analysis.

- *Perforation Output Feature Class*: Name and path of the output file.

Tool 4 – Gained and lost patches

- *Landscape Before*: Polygonal FC of the landscape in the earliest moment of analysis.

- *Landscape After*: Polygonal FC of the landscape in the latest moment of analysis.

- *Gained Area Output Feature Class:* Name and path of the output file.

- *Lost Area Output Feature Cass:* Name and path of the output file.

- *Gained Patches Output Feature Class:* Name and path of the output file.

- *Lost Patches Output Feature Cass:* Name and path of the output file.

3.1.3. Demonstration

3.1.3.1. Background

Traditional olive groves are a characteristic element of Mediterranean landscapes. Besides its historical and cultural value, traditional groves often host a multitude of

species, making them relevant in terms of biodiversity. In Portugal, new high-yielding intensive groves have been fast expanding and replacing several other LULC, among them the biodiversity-rich but low-yielding traditional olive groves (Morgado et al. 2020). Two of the most noticeable negative impacts originated by the way the activity has been conducted are the abusive use of agrochemicals close to the villages and the nocturnal mechanical harvesting that leads to mass bird mortality (Silva and Mata 2019). This analysis, however, focuses on the major land transformations that can lead to landscape homogenization provoked by large-scale plantations. Detecting and identifying the recent olive groves spatial pattern dynamics and being able to make scenarios can contribute to minimize the negative environmental impacts of the modern groves, capable of high yield production.

3.1.3.2. Material and methods

A File Geodatabase was built using ArcCatalog and the FC were imported into it. Next, a blank ArcMap project was created, and both the current workspace and the scratch workspace were pointed to the built geodatabase. Then the LDTtool toolbox was added to the ArcToolbox and the FC were loaded into the project.

We used the mainland Portuguese Administrative Boundaries Official Map (2018 version) as study area and the Olive Groves of 1990 and 2018 extracted from the CORINE Land Cover (CLC) maps. The analytical square size was defined as 10 km and the minimum size patch (to discard spurious polygons resulting from intersect operations) as 50.000 m^2. It is important the dimensions are appropriated relative to both the base maps and the phenomena under study. Since CLC uses a minimum size patch of 25 hectares (250.000 m^2), selecting smaller minimum size patches does not to improve the spatial resolution. Also, we knew beforehand the landscape changes in Portugal involving olive groves are primarily due to large scale plantations, often larger than 50.000 m^2.

The output FC was named "LDT_OliveGroves" (Figure 9a). This FC was then used as input for 'Tool 2 – Forecast' (Figure 9b).
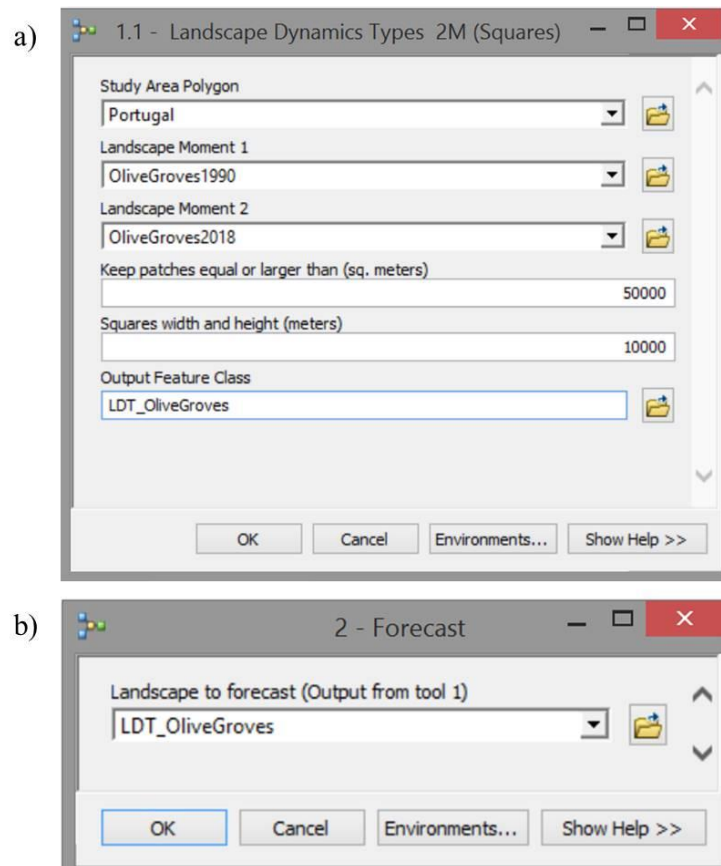
Figure 9 – Filled dialog boxes. a) Tool 1.1 – Landscape dynamic Types 2M (Squares); b) Tool 2 – Forecast.

3.1.3.2.1. Square size selection – Tackling MAUP

The LDTtool allows the use of districts and squares as AU. Commonly used in population-focused studies, data aggregation is often based on boundaries such as zip codes, census tracts and census block groups (Moon and Farmer 2001). Regular grids are extensively applied in spatial analysis to reduce bias caused by administrative divisions (Swift et al. 2008) and/or to include the vast amount of data that is available in that format, such as remote sensing data, biological atlas or simply because there is a preference for working with squares (e.g., UTM) or hexagons as a reference. The use of an artificial sampling grid requires caution to make sure the sampling scheme itself is not adulterating the raw data. Since there is not a single solution for this problem and there are a multitude of contexts and applications, users often must find a way to assess

quality and establish tolerance levels. One important aspect to consider is how well the sampling scheme preserves the integrity of the base data. For instance, in our case study, a single patch can be recognized by the software as being multiple patches because it crosses several squares (or districts). Since the final calculations are based on the AU and not on the patches themselves, the same patch can be counted multiple times. A good compromise would be a grid size that minimizes patch intersection and still offers an adequate spatial resolution to assess the phenomenon being studied. The olive groves maps show 1.613 patches in 1990 and 1.974 in 2018. After intersecting the grid composed by 1.011 100 km^2 squares, the NP was 2.220 and 2.685, respectively. This represents 607 artificial extra patches for 1990 and 711 for 2018. The intersections per square were 2,2 for 1990 and 2,7 for 2018. To increase the spatial resolution requires smaller squares that would make these numbers higher, and therefore worse. On the opposite direction, to lower these numbers we would have to enlarge the squares and lose interpretative spatial resolution.

As to patch size, the average olive grove was 1,72 km^2 in 1990 and 1,76 km^2 in 2018. Thus, a 100 km^2 square would be large enough to fit several patches and to minimize the original patches crossing several squares. Only one patch in 1990 and 2018 is larger than 100 km^2, which suggests that the grid size is not too small. It could be larger but the larger the square is, the more patches it can fit and that increases the risk of hiding some ToD by losing spatial resolution. Overall, these values seemed acceptable for the task in hands, but could the square size influence the output too much? To answer this question, we ran the tool using 8 km, 9 km, 10 km, 11 km and 12 km squares that differ 10% and 20% from the original 10 km grid. This allowed us to assess the MAUP's scale effect. Having different sizes, the grids are distinct, the squares are not in the same place, and with that we were also assessing the MAUP's context or zoning effect. With the 10 km x 10 km squares as a starting point, we used R software (R Core Team 2018) to run Chi-Squared tests against all the other grid sizes, based on the ToD counts (Table 3).

Table 3 – Number of squares with each type of dynamic (ToD) in different size squares.

ToD	Square size (km x km)				
	8	9	10	11	12
A	9	11	8	6	7
D	76	68	49	43	34
E	116	95	71	61	39
F	296	239	227	202	174
G	26	25	16	16	15
H	153	102	101	77	74
I	60	66	55	47	49
Absent	803	627	484	389	327

3.1.3.3. Results and discussion

All the Chi-squared tests had the same result: Do not reject the null hypothesis that assumed the values are similar / equivalent (Box 1). Knowing that no significant difference existed between the results (in terms of ToD counts) we continued the analysis using the 10 km x 10 km squares.

Box 1 – Chi-squared tests for squares with different sizes.

```
maup_10vs8<-table(type,size)                    maup_10vs11<-table(type,size)
maup_10vs8                                      maup_10vs11
chisq.test(table(type,size))                    chisq.test(table(type,size))
X-squared = 9,2028; df = 7; p-value = 0,2384    X-squared = 1,5007; df = 7; p-value = 0,9823


maup_10vs9<-table(type,size)                     maup_10vs12<-table(type,size)
maup_10vs9                                       maup_10vs12
chisq.test(table(type,size))                     chisq.test(table(type,size))
X-squared = 6,8288; df = 7; p-value = 0,4469     X-squared = 4,8826; df = 7; p-value = 0,6743
```

A complementary visual analysis of the ToD spatial distribution for different square sizes also reveals that despite some variation, the major patterns are recognizable in all maps (Figure 10).
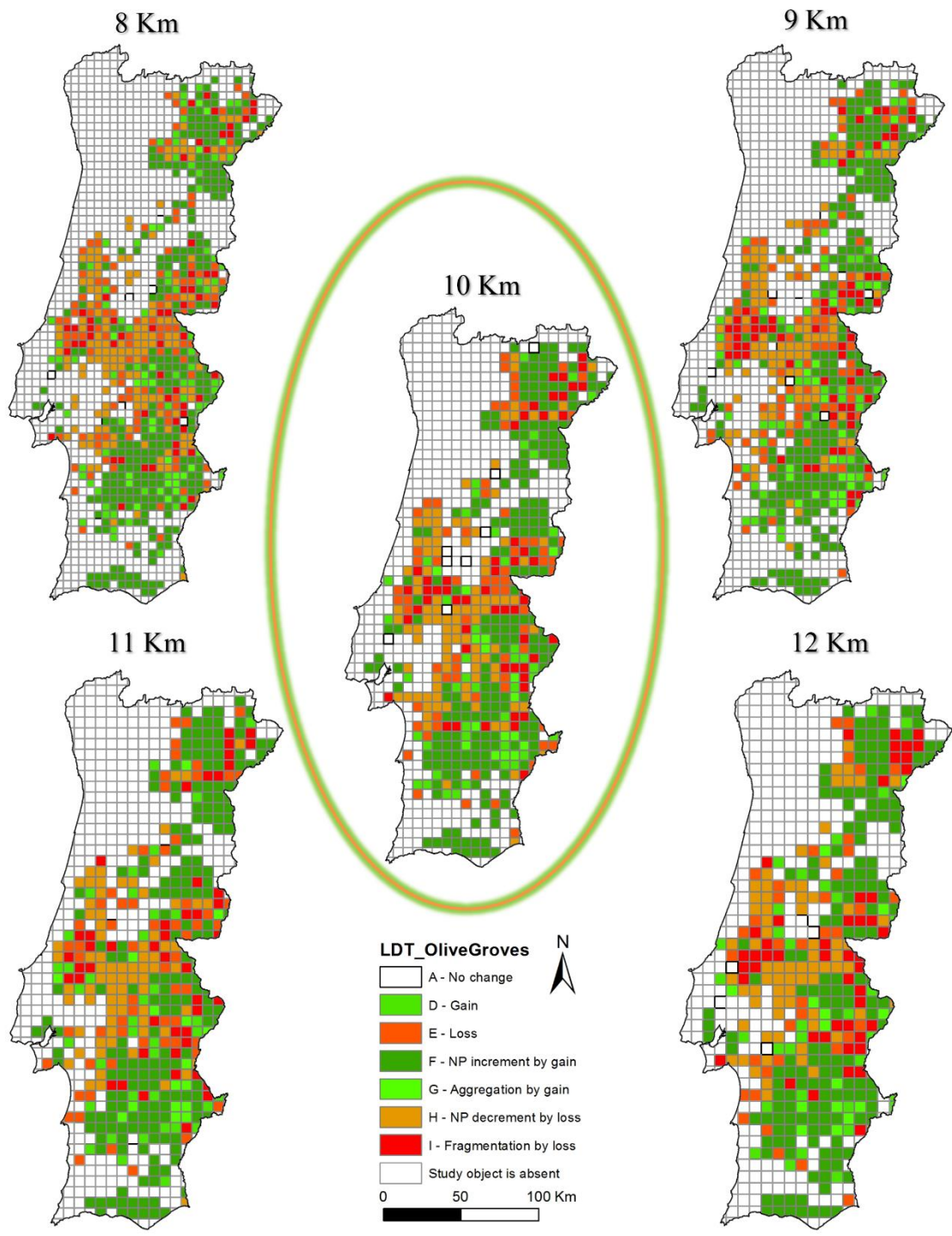
Figure 10 – Maps of types of dynamics for squares with different sizes.

The outcome is a FC containing the squares that comprise the study area, each one with the metrics calculated for both dates, their variations and assigned to a ToD (Figure 11 and Figure 12).

Figure 11 – Output attribute table. a) Output of Tool 1.1 - Landscape dynamic Types 2M (Squares); b) Field originated by Tool 2 – Forecast.
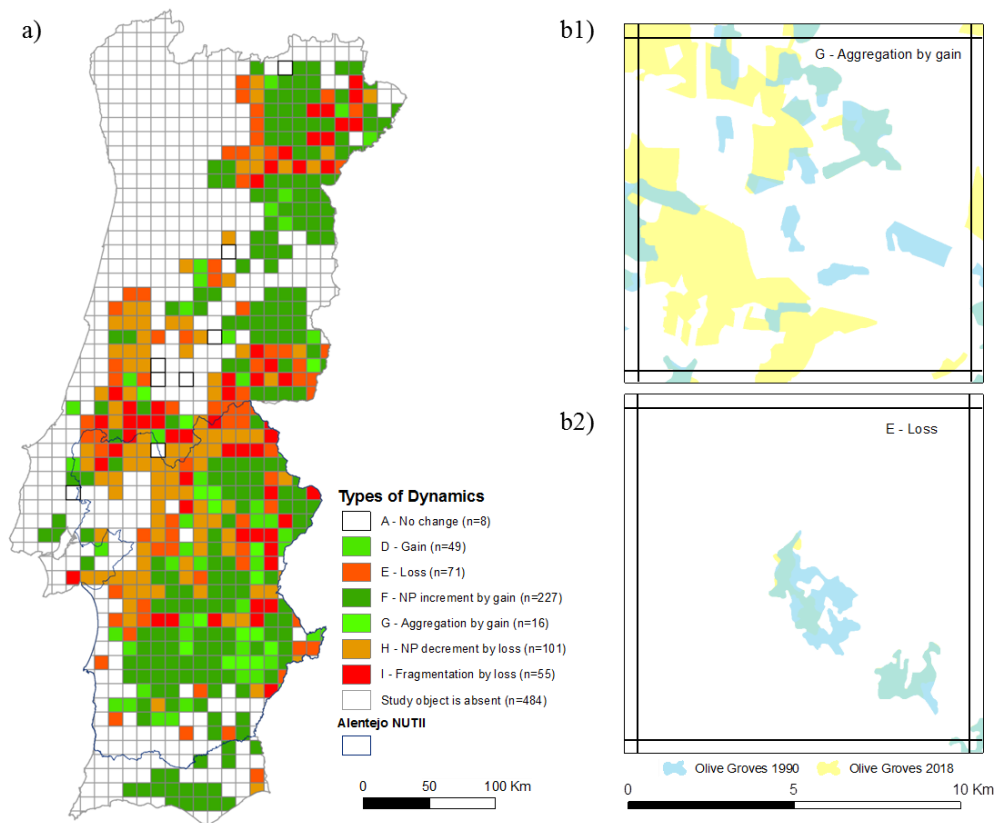
Figure 12 – Olive Groves in Portugal (1990-2018); a) Map produced by "Tool 1 - Landscape Dynamics Types 2M (Squares)" showing mainland Portugal divided by 10 km x 10 km squares with Types of Dynamics (ToD). b1) Detailed view of a square showing TOD G - Aggregation by gain. b2) Detailed view of a square showing ToD E – Loss.

Although other drivers may be responsible for local changes occurring all over the country, it is hard to ignore the process of rapid agricultural intensification going on in the Alentejo Region (Southern Portugal) and its relevance to the landscape

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transformation. The traditional extensive, multi-functional agricultural systems, adapted to the Mediterranean climate are being replaced by more intensive and irrigated cultures. This was made possible due to public investment in the Alqueva dam and its integrated irrigation system, together with national and European agricultural policies (Silveira et al. 2018). As a result, large scale intensive and super-intensive olive groves have been implemented in the last years in the region. According to CLC (CHA00, CHA06, CHA12 and CHA1218) the land covers that have been replaced the most by olive groves are 'non-irrigated arable land' and 'permanently irrigated land', presumably land that was not irrigated before but has now access to water, and land that was already irrigated but was used for other cultures. Morgado et al. (2022), studying the municipalities holding most olive farms in Alentejo, reported a large-scale expansion of irrigated olive groves between 1990 and 2017, made mostly by replacing open dry farmland (63%) and traditional groves (21%).

We verify an olive grove area increment of 692,84 km^2 and a NP increment of 361 between 1990 and 2018. These overall values suggest the 'ToD F – NP increment by gain' has been the main change pattern taking place at the study area. The results based on the 100 km^2 grid provide more detailed results. The number of squares assigned to each ToD is present in Figure 13a).
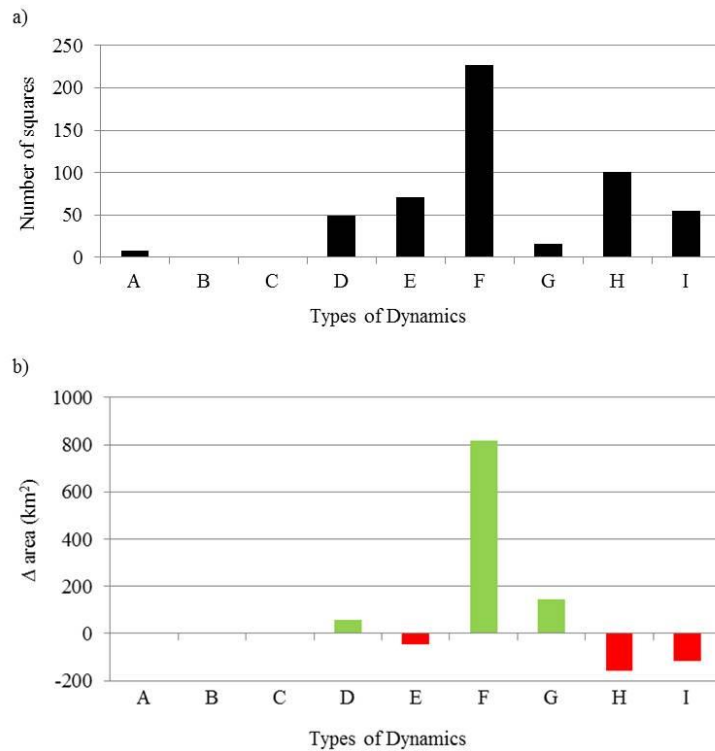
Figure 13 - a) Number of squares assigned to each Type of Dynamics; b) Olive grove area variation associated to each Type of Dynamics.

The 'ToD F - NP increment by gain' was the most represented with a total of 227 squares which is in line with the global trend previously mentioned. It represents new olive groves not adjacent to existing ones (new patches) and the magnitude of ToD F reflects how vigorous the olive grove implementation has been. The second most identified ToD was the 'H - NP decrement by loss', present in 101 squares. It shows squares where the main dynamic was the shift from olive groves into other land covers. Looking at the number of squares it seems quite significant with almost half of the ToD F (101 vs 227). However, the relation does not stand when we compare olive grove area variation as ToD F involved an area gain of 817,57 km^2 while ToD H involved an area loss of 159,61 km^2 (Figure 13b). ToD H occurred on several locations but did not involve large amounts.

The other two ToD with variation in both configuration and composition are 'G – Aggregation by gain' that reflects situations where new plantations or expansion of existing ones originated the fusion of patches (groves), and 'I – Fragmentation by loss', where the loss of area provoked the division of patches. ToD G was identified in only 16

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squares but totaled an area gain of 143,82 km^2, while ToD I was identified in 55 squares and implied an area loss of 118,05 km^2. Focusing on the ToDs that reflected only composition and had no influence in the NP, 'D – Gain' was found in 49 squares and involved an area gain of 57,18 km^2, while 'E – Loss' occurred in 71 squares with a total loss of 47,83 km^2. No squares were assigned to the ToD B or C, that represent pure geometric changes (without amount variation). Overall, the olive grove area increased despite some local losses and most of the amount changes had geometric implications.

The LDT forecast scheme allows us to see that if the olive groves area keep increasing, in time, the existing groves may expand so that they bridge the gaps between them and merge together (ToD D evolves to ToD G). New groves may follow the same path (ToD F is followed by ToD D which evolves to ToD G). It is important to highlight this is a theoretical simulation, useful to scenarize and compare alternatives rather than a prediction, that would require additional data and several variables, such as natural impediments to the trend and the importance of human intervention. Olive groves will not expand by natural dispersal; instead, they are highly managed, and their dynamics are almost exclusively dependent on human decision. For instance, Morgado et al. (2022) pointed out the joint effect of farm size and inclusion in public irrigation systems as key factors behind the transition to irrigated olive groves in the period 1990–2017. Identifying the drivers of LULC change is beyond the scope and aim of this analysis which focuses on detecting major spatial patterns. However, more thorough results are possible to obtain using LDTtool if the inputs have higher spatial, temporal (to search for different pace in the changes), and thematic (to distinguish between traditional rainfed and modern irrigated olive groves) resolutions.

3.1.4. Conclusions

The LDTtool can add value to studies in a wide range of topics related to biodiversity conservation, invasive species, ecosystem services and natural resources planning, mainly if LULC (or habitat) changes are a key aspect. The LDTtool is an ArcGIS toolbox and therefore is expected to be user-friendly. The configuration is straightforward, and it runs smoothly and fast mostly because it uses the vectorial format. It is implemented

in Python, which is a versatile language widely used in geoprocessing and additionally the source-code is available allowing users to modify it to their needs. The possibility to adjust parameters such as the minimum patch size and the AU also add versatility to the process. The fact that it uses binary landscapes may be considered a weakness as it may oversimplify the reality for certain types of studies such as those involving habitat suitability or functional connectivity assessed by resistance surfaces.

Besides its usefulness and user-friendliness, LDTtool represents an integrated and methodologically supported vision of the whole process (concept-method-tool). As a tool, LDTtool is not a final product; it was already updated and enhanced since its inception and will continue to be so. The original version released together with the paper "LDTtool: A toolbox to assess landscape dynamics" (Machado et al. 2020) contained only four LDT core tools and the 'Forecast' tool. The 'ToD A1 – Spatial Shift', the 'ToD E1 – Perforation', as well as the tools 'Perforation' and 'Gained and Lost Patches' were integrated later. One potential improvement that was identified in an early stage is the possibility to expand the analysis beyond binary landscapes. However, the most relevant step to increase the tool's usability has already been taken and is the topic of the next subchapter: integration in other Geographic Information System (GIS) platforms.

3.2. LDT4QGIS: a free and open-source tool to assess landscape dynamics.

This subchapter is based on the article:

Paixão, L & Machado R. LDT4QGIS: a tool to enhance landscape analysis in QGIS. (*submitted*)

Authors' contributions:

Luís Paixão: Software, Validation, Writing – Review and Editing; Rui Machado: Conceptualization, Methods, Testing, Writing the original draft.

3.2.1. Background

The conceptual framework and nomenclature establishment (subchapter 1.6) paved the way for a method development (LDT) (Chapter 2), automated in ArcGIS by a tool (LDTtool) (Chapter 3.1.). This subchapter introduces LDT4QGIS, a similar tool for QGIS. Most overall traits remain like the use of vectorial and not raster format as in most landscape analysis software. The raster format is particularly useful for storing data that varies continuously (e.g., temperature, elevation, etc.), which is not the case here. For land cover maps in which thematic classes needs to be distinguished and limited by boundaries, the choice is between a discrete raster, that consists of integers used to represent classes, and a vectorial data model. However, given that LDT4QGIS uses binary landscapes (only one land cover category is geoprocessed) and considering that the typical raster advantages no longer apply, it makes sense to use a vectorial format since it may provide more geographic accuracy, as it is not dependent on the pixel size, and usually performs better in terms of data storage and processing speed. Besides these operational aspects, the use of a vectorial format is also convenient since most official land cover maps are available to the public in vectorial format.

However, the most distinctive feature of this method and associated tools is the fact that it goes beyond metric calculations and focuses on the spatial processes in land transformation. Metrics and indices are undoubtedly useful to describe landscapes and

their changes, but since certain variations in particular metrics reflect determined spatial processes, it makes sense to go one step further and provide those processes as outputs of the analysis.

3.2.2. LDT4QGIS description

LDT4QGIS consists of three python scripts that allow the application of the LDT method in QGIS.

- Script 1 - "LDT4QGIS.py": This is the main script that performs the LDT steps to identify the ToD in the landscape. The analysis can be conducted using two or three analytic moments (2M or 3M) and be spatially represented in a regular vectorial grid built during the analysis (Squares) or in an irregular polygonal study area provided by the user (Districts). There is also an option to include 'E1 – Perforation' as an assignable ToD to the AUs.
  - Landscape Dynamic Types 2M (Squares)
  - Landscape Dynamic Types 3M (Squares)
  - Landscape Dynamic Types 2M (Districts)
  - Landscape Dynamic Types 3M (Districts)

- Script 2 - "perforation.py": This is an accessory script to geoprocess, at the class scale, the spatial pattern 'perforation'. It shows where perforation happened between two analytical moments. Within the LDT framework, it is accurate to state that perforation is a particular case of 'ToD E – Loss' in which the lost amount originates a clearing.

- Script 3 - "gained and lost patches.py": This is an accessory script to identify and locate the places where amounts of the LULC category of interest were gained and lost between two analytical moments, including new individual patches or individual patches that disappeared. It produces four output shapefiles containing all gained areas, all lost areas, gained patches and lost patches.

3.2.2.1. Preliminary steps, inputs and settings

To prevent errors or malfunctioning, all the inputs elements should share the same CRS and avoid unnecessary fields in their attribute tables. The outputs are only as good as the inputs that generate them, therefore proper spatial and temporal resolutions are required to correctly assess landscape changes. The scripts are simple to use but some details regarding data pre-processing are worth mentioning:

1. A projected CRS should be used, and the coordinates should be displayed in meters.
2. The input land cover shapefiles must contain only one category with the polygons of interest. Hence, depending on the base maps available, it may be necessary to export the category under study to a new shapefile and use it in the analysis.
3. Scripts should be stored in the QGIS script folder. The path to the folder is shown in: Processing Toolbox – Options – Processing – Scripts (Figure 14). Exiting and restarting QGIS will ensure that the scripts are automatically loaded to the Processing Toolbox.
4. Two symbology files (suited for 2 or 3 analytical moments) are provided. If stored in the same folder along with the scripts the symbology is automatically applied.
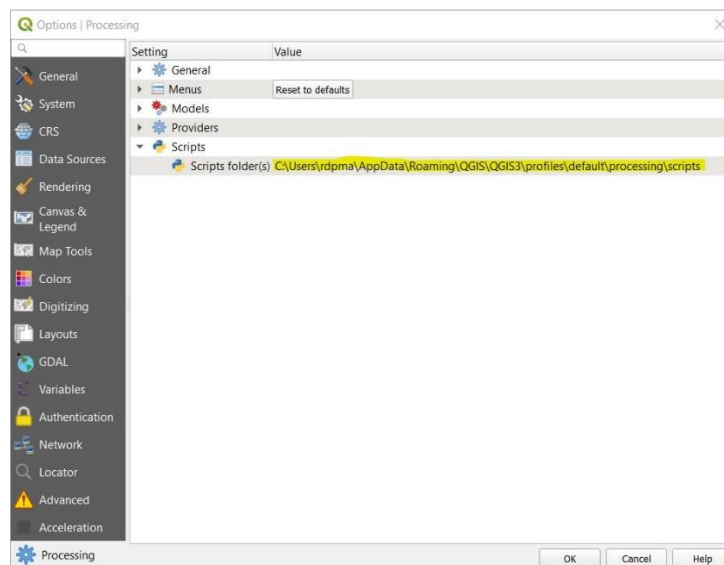
Figure 14 - Path to the QGIS script folder (Processing Toolbox – Options – Processing – Scrips).

The inputs and settings required to run the scripts are the following:

Script 1 - "LDT4QGIS.py"

• *Moments:* Choose whether the analysis should be based on two or three moments.

- *Type of Analysis*: Choose whether the analytical units should be squares or districts (Polygonal shapefile containing the districts' boundaries, provided by the user.)

- *Study Area Polygon*: Provide a polygonal shapefile containing the study area boundaries.

- *Landscape Moment 1*: Provide a polygonal shapefile of the landscape in moment 1.

- *Landscape Moment 2*: Provide a polygonal shapefile of the landscape in moment 2.

- *Landscape Moment 3*: Provide a polygonal shapefile of the landscape in moment 3.

- *Keep patches equal or larger than (square meters)*: Choose the minimum patch size to be analysed.

- *Squares width and height (meters):* Choose the analytic square size (for a square-based analysis only).

- *Perforation:* Check the box to include the 'ToD E1 – Perforation' to the analysis.

- *Output Shapefile*: Provide the name and path of the output file.

Script 2 - "perforation.py"

- *Landscape Before:* Provide a polygonal shapefile of the landscape in the earliest moment of analysis.

- *Landscape After:* Provide a polygonal shapefile of the landscape in the latest moment of analysis.

- *Output Shapefile:* Provide the name and path of the output file.

Script 3 - "gained and lost patches.py":

- *Landscape Before:* Provide a polygonal shapefile of the landscape in the earliest moment of analysis.

- *Landscape After:* Provide a polygonal shapefile of the landscape in the latest moment of analysis.

- *Output Path:* Select the folder in which the output shapefiles will be saved.

- *Output Name:* Select a prefix to the output file names. The names are by default "GainedPatches", "LostPatches", "GainedArea" and "LostArea".

3.2.3. Examples

3.2.3.1. Grasslands in Slovenia

European semi-natural grasslands are often considered to have high conservation value due to the unique and highly diverse communities of species they harbor (Veen et al. 2009). However, extensive grasslands figure among the most endangered ecosystems in the European Union (European Environment Agency 2020). Marginal areas are prone to be abandoned and face problems of overgrowth, and the best lands are frequently used for more intense agriculture, often associated with increased fertilization, mowing frequency and other practices associated with the deterioration of grasslands (Stoate et al. 2009; Van Vooren et al. 2018). Some polices, particularly grassland-specific agri-environmental measures, have been implemented in Slovenia, but the outcomes were far from expected (Kaligarič et al. 2019). Better and more informed policies need to be designed to prevent further degradation and loss of grasslands. Some grasslands are probably lost forever, and others are experiencing an accelerated pace of degradation; therefore, it is fundamental to understand the recent trends, model future LULC changes, and prioritise which grasslands are most important and worth preserving. This example consists of applying LDT4QGIS to Slovenian grasslands using 2.500 ha (5 km x 5 km) squares. The base datasets are the official landscape land cover maps (https://rkg.gov.si/vstop/) for the years 2009 and 2020 (Figure 15).
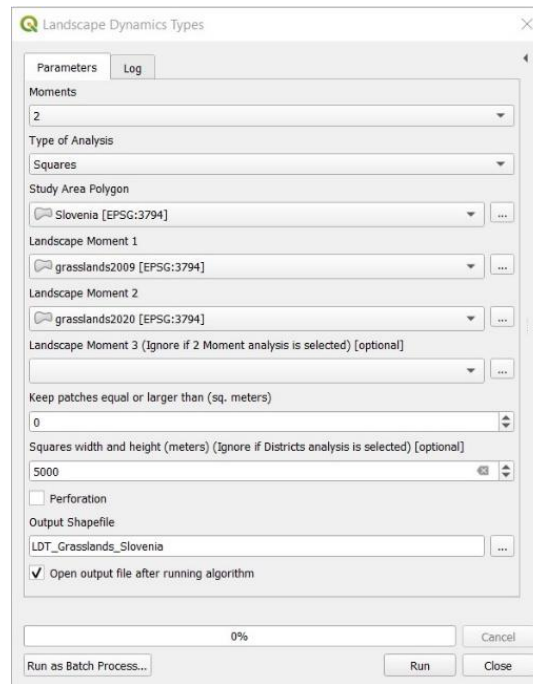
Figure 15 – Example of inputs and settings for the tool 'Landscape Dynamic Types 2M (Squares)'.

The results reveal dominance of squares related to loss (ToD E, H and I) compared to those related to gains (ToD D, F and G) (Figure 16). Squares with ToD I represent areas where one or more grassland patches lost area in a way that patches were divided, increasing the total NP. That happened nationwide but mainly in the centre of Slovenia, around the capital, Ljubljana. Squares with ToD H reflect areas where one or more grassland patches disappeared; ToD H can be interpreted as the phase that follows ToD I. For reference, in Forman's framework, this would equate to fragmentation followed by attrition, possibly with shrinkage as an intermediary phase (Forman 1995a). The process initiated with an area loss that caused the division of large patches, and then smaller patches began to disappear. Given that ToD I is the precursor of ToD H, if the declining trend continues, it is expected that some (or many) squares now identified as ToD I will evolve to ToD H.

The loss of grasslands represents direct habitat loss for species that highly depend on them (specialists). The main spatial pattern in which the area loss occurs (ToD 'I – Fragmentation by loss') is also likely to be especially detrimental for specialists because it promotes the breaking up of patches into smaller patches, bringing more landscape heterogeneity, more edge effect, and overall higher levels of disturbance. For generalist

species, intermediary levels of landscape fragmentation are often acceptable and even beneficial, but if the loss of grasslands continues and triggers ToD H, it would most likely represent landscape homogenization (with other composition different than grasslands) and therefore a less suitable habitat for some generalists.

Ultimately, conservation goals are set based on biodiversity values and experts' opinions and the results produced by LDT4QGIS can be combined with other sectorial and thematic data to produce more explanatory information. Future projections based on current trends provide fundamental insights to design better land-use policies and concrete measures to assure a balanced coexistence of agricultural practices and biodiversity conservation.
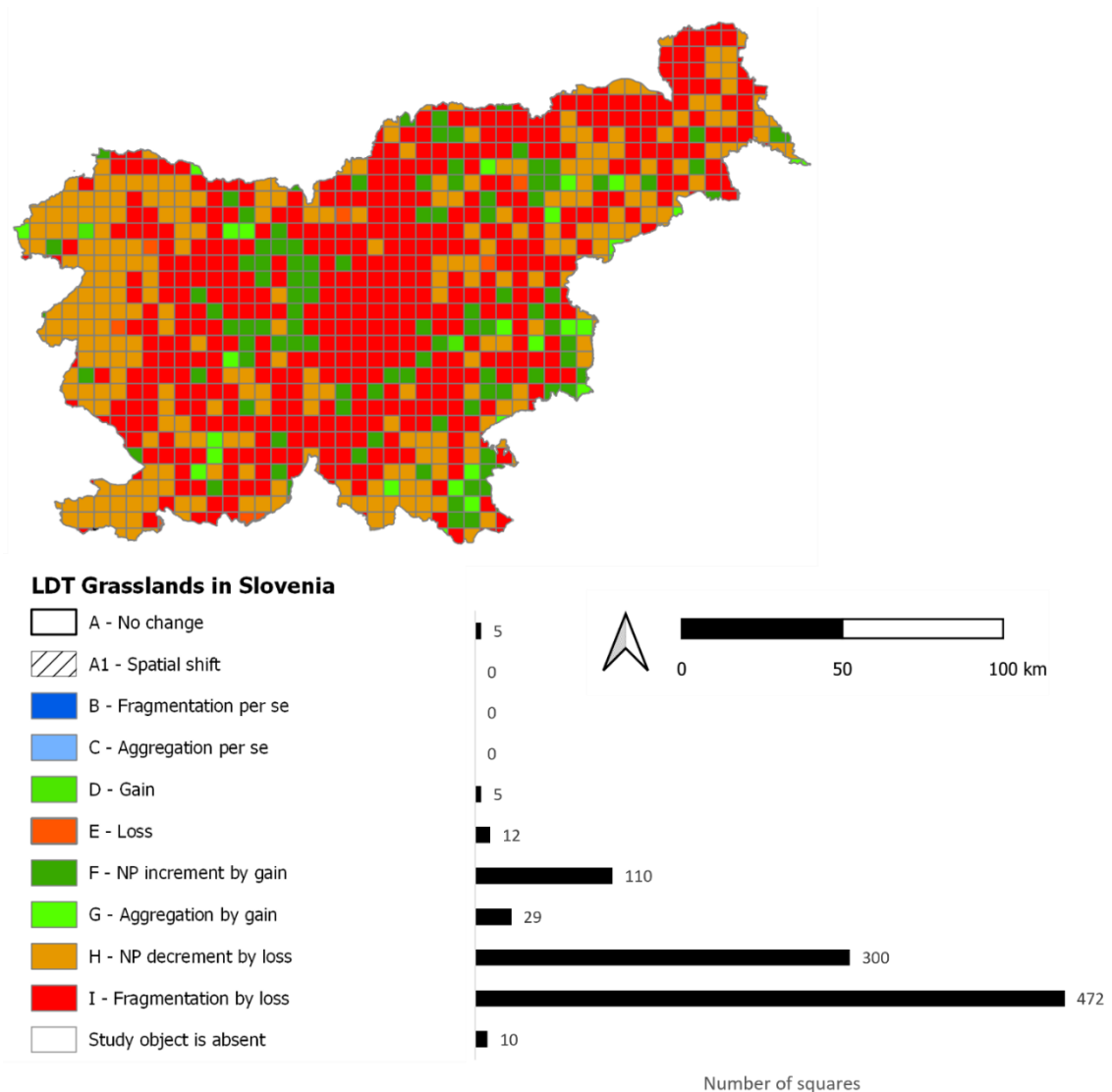
Figure 16 – Map of the case study regarding the grasslands in Slovenia with associated legend to identify the types of dynamics and chart showing the number of squares assigned to each type.

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3.2.3.2. Olive groves in Portugal

Traditional olive groves are an historical element of Mediterranean landscapes. Besides the sense of belonging derived from the rural culture acquired by several generations, olive groves provide food and wood. The balanced management typical of the traditional groves makes them relevant as far as biodiversity is concerned since they harbour numerous fauna and flora species (Zaccarelli et al. 2008; Fernández-Habas et al. 2018). In the Iberian Peninsula, the quest for higher yields has led to a noticeable landscape change over the last few decades. In Portugal, high-yield irrigated olive groves have been fast expanding often replacing traditional biodiversity-rich groves and other land-use types (Morgado et al. 2020, 2022). The main challenge today is to successfully implement the more intensive and economically relevant production models while mitigating the negative environmental impacts of these large-scale industrial groves.

To address this topic, it is fundamental to assess olive groves changes, monitor their expansion patterns and determine how these dynamics are contributing to landscape homogenization. In this example, to differ from the previous example and demonstrate a different LDT4QGIS feature, the analysis is conducted based on administrative boundaries, using the smallest Portuguese administrative unit *freguesia* (parish). The data used are the Portuguese official land-use map (Carta de Uso e Ocupação do Solo) for the years 1990, 2010 and 2018 (Direção-Geral do Território 2019) (Figure 17).
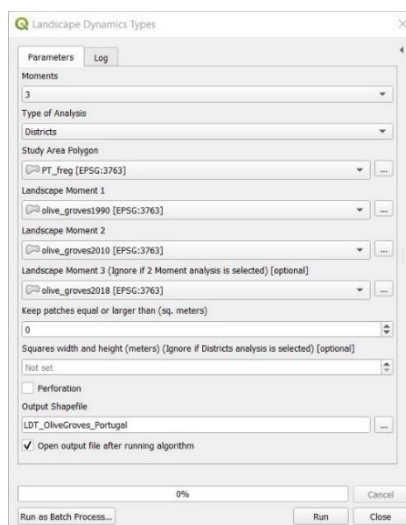
Figure 17 – Example of inputs and settings for the tool 'Landscape Dynamic Types 3M (Districts)'.

The analysis involved three moments, so three maps were produced: one with the dynamics between 1990 and 2000, one between 2000 and 2018, and the one displayed in Figure 18, which shows the overall changes between 1990 and 2018. We can see that olive groves are not present in most mainland Portugal parishes. There are 383 parishes with ToD related to gain and 376 related to loss, but the magnitude of the transitions is quite different, with an overall gain of 69.493 hectares. Focusing on the gain side, 275 parishes display the 'ToD F – NP increment by gain', which reflects the emergence of new groves isolated from the existing ones. 'ToD D – Gain', which identifies areas where existing groves increased their size, was assigned to 77 parishes. This suggests that, at the parish scale, the increase in area is mostly obtained via new plantations rather than through the expansion of existing groves. Finally, 31 parishes increased the total area covered by olive groves area while decreasing the NP ('ToD G – Aggregation by gain'). Although olive groves have been increasing significantly, they still represent a small percentage of agricultural land, and thus the identification of 31 parishes with ToD G seems reasonable, but further area gains will increase the probability of the occurrence of this ToD. The area covered by olive groves and other intensive perennial crops is expected to increase in the future, pushed by the increasing popularity of Mediterranean products (Esgalhado et al. 2021), and although further investment may be desired due to the positive impacts on production, the extension and spatial context of such plantations should be carefully examined.

Outputs such as those obtained through LDT4QGIS are useful for diagnosing the current situation, as well as for simulating what different land transformation alternatives could represent in terms of spatial patterns and the associated environmental impacts. Such elements, combined with measurable indicators or decisive criteria, can help policymakers to predict the effects of agricultural policies in the territory. Some decisive criteria could be: 'preferentially place olive groves in areas highly dominated by other LULC', 'do not promote patch fusion (ToD G) if olive groves cover more than 40% of the parish, square, or defined proximity neighbourhood', 'establish a maximum plantation size, unless the perimeter-area ratio is ≥ …', and 'preserve or restore x ha of different LULC for each ha of irrigated cultures above the x ha threshold'.
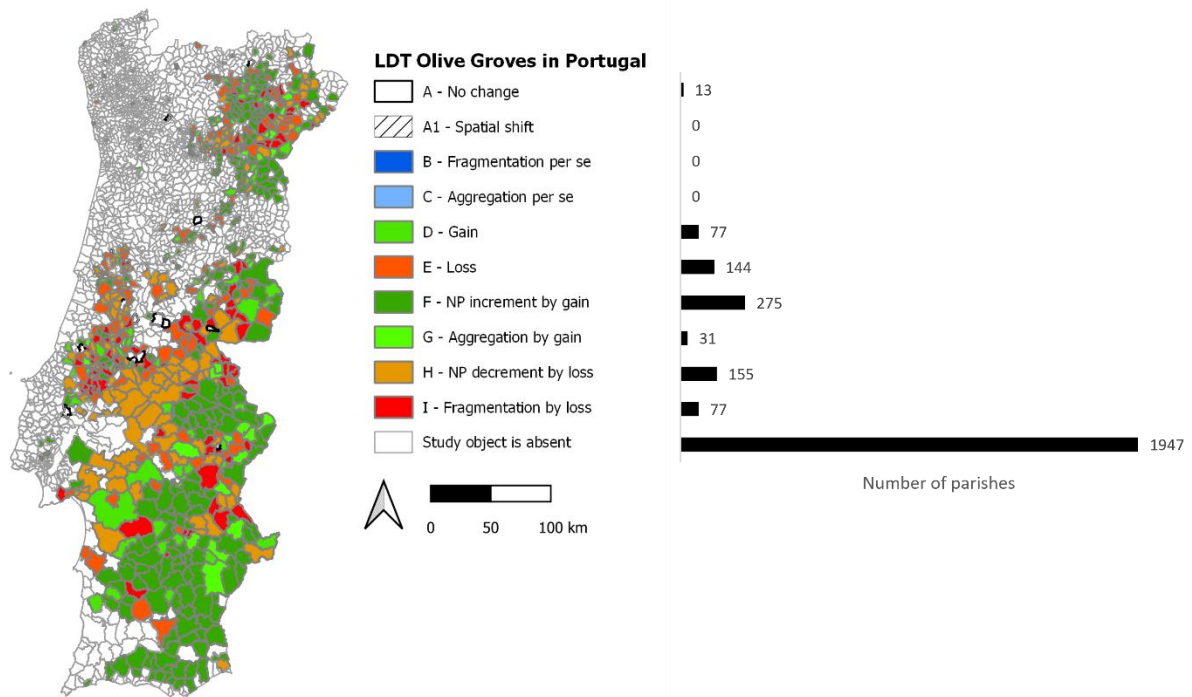
Figure 18 – Map of the case study regarding the olive groves in Portugal with associated legend to identify the types of dynamics between 1990 and 2018, and chart showing the number of parishes assigned to each type.

3.2.3.3. Forests in Germany

Forests provide several services such as water regulation, carbon sequestration, biodiversity conservation, recreation, etc. (Ninan and Inoue 2013) making them systems of the utmost importance. Threats to forests such as wildfires, pests, diseases and how much and where to harvest should always be accounted for and anticipated. In this regard it is useful to assess LULC transitions involving forests, whether purely amount-related (how many hectares lost or gained), or also considering geometric features (e.g., changes comprising patches entirely or partially.) For instance, the spatial process known as 'perforation', which involves removing (losing) a given amount inside a patch originating a clearing, is a common way to trigger land transformations (Forman 1995a); these changes can be small at first but can escalate and evolve to more severe processes of, in this case, forest loss, fragmentation and overall degradation. In this example, we ran the scripts 2 ('perforation.py') and 3 ('gained and lost patches.py') with CLC maps to

identify all the gains and losses, gained and lost patches and perforations that occurred in the German forests between 2000 and 2018 (Figure 19).
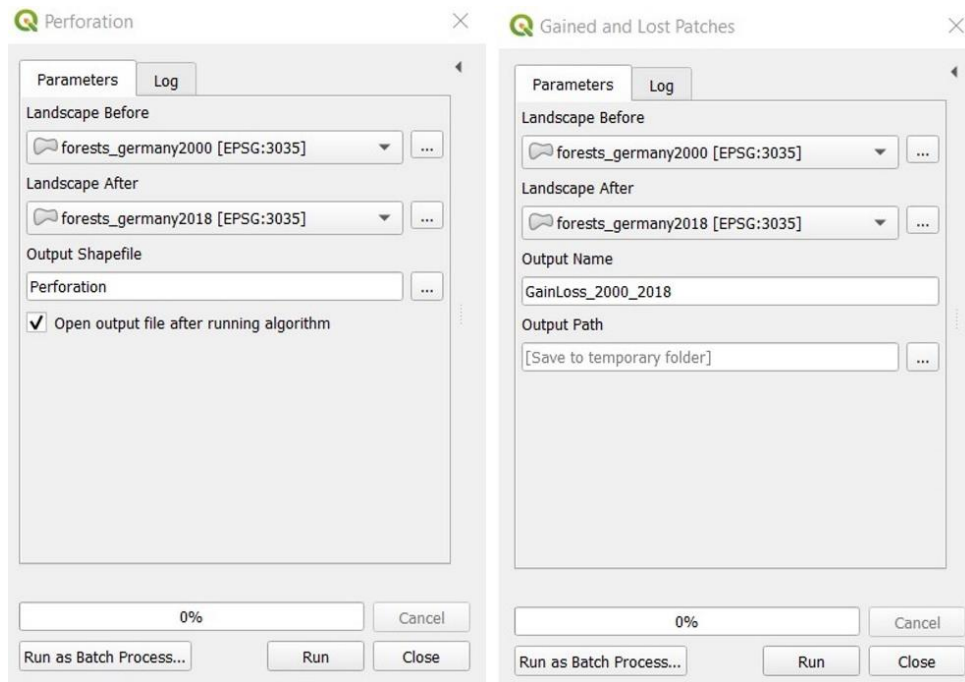
Figure 19 – Example of inputs and settings for the tools 'Perforation' and 'Gained and Lost Patches'.

Between 2000 and 2018, the German forests had a net gain of 420.999 ha (10.383.107 ha in 2000 and 10.804.106 ha in 2018). The partial balances are 1.161.929 ha of new forest and 740.930 ha of lost forest. These numbers show that the area covered by forest has been increasing, and the areas converted from and to forest (the sum of the partial balances = 1.902.859 ha) represent 18% of the average forest area (average of the total areas in 2000 and 2018 = 10.593.607 ha), which suggests that the system is fairly stable.

Going beyond a strictly numerical approach and using the spatially explicit information provided by LDT4QGIS, we can see where these changes occurred throughout the study area. Identifying the gained and lost individual patches is important because the spatial patterns created by forest gains (e.g., natural afforestation or planting) or losses (e.g., burnt by wildfire or harvested) may have major ecological consequences (Franklin and Forman 1987).

Not considering strictly amount variations like 'ToD D – Gain' or 'ToD E – Loss' and looking only to emergence and disappearance of individual patches, we found 22.312 gained patches and 9.066 lost patches. The analysis also showed 936 perforations summing a total of 37.172 ha. Figure 20 shows an overall view of German forests in 2000 on the left-hand side, and an example of perforation on the right-hand side. The light green representing the forests in 2000 can be seen behind the darker green that represents the forests in 2018. Although more areas shown in the image were lost, only the one that was initially completely contained by the forests of 2000 and is now lost, forming a clearing, is classified as perforation (hashed red).

For simplicity, we considered forest as a single LULC category. Forest segmentation by species, age, density, function, or physiological condition would make sense for more detailed approaches, and LDT4QGIS would be able to deliver potentially relevant outputs (e.g., distinct ToDs for primary forests and managed forests in the same AU) concerning the spatial pattern dynamics and their consequences.
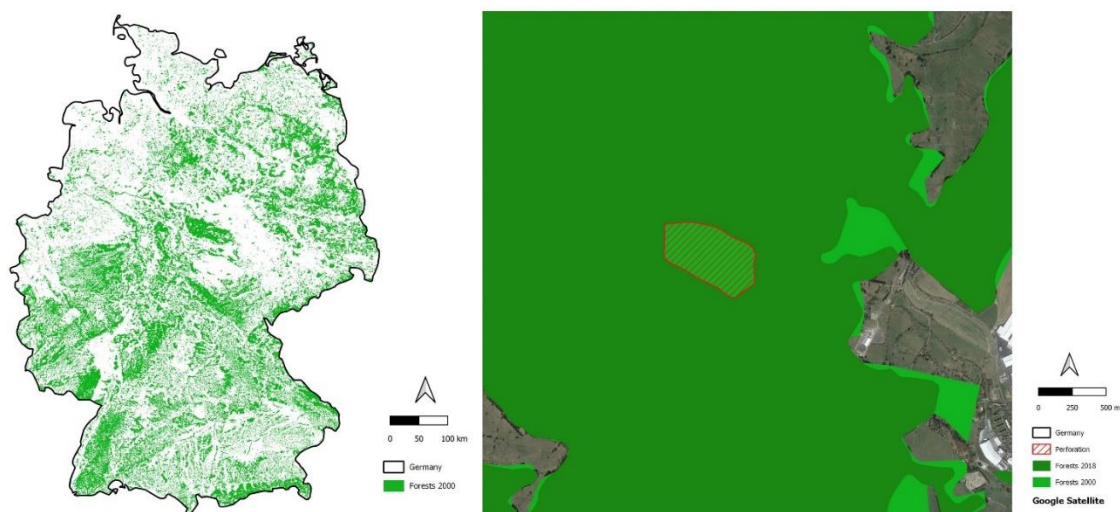
Figure 20 – Map of the German forests in 2000 and a close view of lost areas (light green) and a detected perforation that occurred between 2000 and 2018.

3.2.4. Conclusions

The previous demonstrations show how LDT4QGIS works and the type of outputs it produces. These are illustrative examples, not exhaustive analysis, that are supported

by brief background descriptions and use real data. Therefore, the results are valid, although deeper studies on each topic could render these results incomplete. Depending on the depth of the analysis, LDT4QGIS outputs can be sufficient, or can integrate wider procedures as inputs or variables. For instance, since most official statistics are aggregated according to administrative units, the ToD values of an output shapefile with districts can be exported to such a database and used for statistical analysis together with other relevant and related variables, such as economic or social indicators. Another example would be to use the final LDT4QGIS squares shapefile as a sampling grid in other specialised software tools for landscape metrics, like FRAGSTATS. This would require one to convert the vectorial grid into a raster format, edit the attribute values accordingly, and select the new file to be used as a provided sampling tile. A multiplicity of metrics could be computed, and after some value extraction and table management steps, we would end up with a table containing all the values of the metrics, as well as the ToD.

Despite some variations, LDT4QGIS shares its strengths and weaknesses with LDTtool, and some derive from LDT itself. On the upper side there is the simplicity to use. It is delivered as a ready-to-use set of scripts, the inputs often do not require much preliminary work and the outputs are very straightforward (values variations and maps). The possibility of fine-tuning parameters and code customization adds versatility to the tool. Moreover, although LDT4QGIS operates based on metric calculations, these are instrumental to the end goal of identifying ToDs / spatial patterns, which are real-world combinations of landscape metrics. The main weakness or feature that may require especial caution is the use of AU due to the MAUP.

By facilitating the identification and geopositioning of ToDs, LDT4QGIS adds value to landscape assessments related to LULC changes. Additionally, LDT4QGIS goes beyond summaries and statistics and provides spatially explicit information. These assessments may encompass topics as varied as biodiversity conservation, ecosystem services, game management, or invasive species control. Quantitative measurements, together with spatially explicit maps, are essential for landscape status assessments and ultimately for an informed policy design.

Compared to LDTtool, LDT4QGIS represents an advance regarding accessibility since it was developed for non-proprietary software, thus fitting a geospatial free and open-

source software philosophy. We expect the development and updating of LDT4QGIS to be ongoing work. In addition to improving the current QGIS tool, we aim to extend these analytical capabilities to other platforms. The growing popularity of R (www.r-project.org) as an open-source computing environment has made it our highest priority regarding the expansion of the tool.

This chapter covers studies that used the LDT principles and some of the tools developed. Subchapter 4.1. focuses on a study about invasive plants management in a Portuguese Natura 2000 Network site. Subchapter 4.2. presents a study case regarding the grassland dynamics in Slovenia. In subchapter 4.3. the decline of the Portuguese *montado* is analysed. Finally, subchapter 4.4. summarizes works conducted by other authors using these tools; one in Romania and one in Indonesia, both using LDTtool in urban growth-related studies, and one in Spain focused on afforestation.

4.1. Invasive Species Management

This subchapter is based on the article published in *Restoration Ecology* as:

Authors' contributions:

Rui Machado: Conceptualization, Methods, Analysis, Writing the original draft; Liliana Neto Duarte: Topic Expertise, Review; Artur Gil: Conceptualization, Review; Nuno de Sousa-Neves, Janez Pirnat, Pedro Santos: Review and Supervising.

4.1.1. Background

Invasive alien species (IAS) rank in the top five direct drivers of change in nature (IPBES 2019), being responsible for numerous negative impacts (Kumschick et al. 2015; Schindler et al. 2015; Jones 2017; Diagne et al. 2020). Management of IAS is often difficult and expensive (Diagne et al. 2021), but with proper planning and resources, biological invasions can be managed and mitigated (Pyšek et al. 2020). Invasive alien

plants (IAP) can alter basic ecological processes such as nutrient cycling or the change of soil biota (Marchante et al. 2008; Vilà et al. 2011). They often benefit from competitive mechanisms (Levine et al. 2003) or performance-related traits (Van Kleunen et al. 2010), like long-lived seed banks (Gioria and Pyšek 2015), and disrupt intrinsic interactions among native species (Callaway and Aschehoug 2000).

It is crucial to define 'success' for each intervention and prioritise the species and the areas to act (van Wilgen et al. 2012). According to the current situation and the available resources, the goal can be to control, contain, or eradicate the species (Simberloff et al. 2013). Control aims to reduce the impact and the abundance of an IAS to an acceptable level although not necessarily limiting its range, while containment seeks to limit the spread by acting mostly at the periphery of the species range (Hulme 2006). Therefore, containment is more appropriate for species that disperse slowly and over short distances, and control is more realistic when dealing with larger peripheries typical of long dispersers with higher expansion rates. Eradication is harder to achieve as it implies eliminating all the individuals and viable propagules of a species within the management unit (Parkes and Panetta 2009), and also that the species has not been detected for a period equal to or greater than its seed longevity.

Many interventions see the removal of alien vegetation as the final goal (Vosse et al. 2008), but that alone does not guarantee the natives will come back and restore the ecosystem as it was before the invasion (D'Antonio & Meyerson 2002; Harms & Hiebert 2006). In fact, passive restoration approaches often fail to avoid re-invasions by the same species or secondary invasions by other species, that are able to capitalize on the disturbance caused by the removal operation (Holmes et al. 2020). Furthermore, ecosystem recovery may fail due to the IAP legacy, in the form of reduced biodiversity, massive seed banks, altered soil chemistry, etc. (Corbin & D'Antonio 2012). To achieve an effective recovery the elimination of the invaders must be complemented by strategies to overcome their legacies (Konlechner et al. 2015). Such active approaches to restoration that are more complex and may involve revegetation with native plants, can be more expensive at the start but tend to deliver better results in the long run (Gaertner et al. 2012).

Because restoring invaded areas is costly, the restoration should be thoroughly designed to prevent poor quality implementation that results in the need for larger

budgets over time (Cheney et al. 2019). A variety of important factors for IAP management are included in comprehensive approaches or procedures introduced by several authors. Higgins et al. (2000) developed a broad conceptual model that includes parameters such as spread patterns and the time needed for the eradication. To account for uncertainty in the analytical procedures regarding the management of woody IAPs, Roura-Pascual et al. (2010) developed a framework for a spatially-explicit sensitivity analysis including factors related to fire risk. Krug et al. (2010) tested budget scenarios and the related efficiency for better prioritize cleaning areas. Prioritization could be conducted according to distinct criteria, from biological to economical or logistical. One way to help optimize recovery and reduce the costs of restoration is to prioritize patches with higher spontaneous succession potential. In terms of logistics, it may be easier to have an entire team working on a large patch than having the elements dispersed in the territory and thus increasing the need for transportation, gear redundancy, etc.

It is appealing to remove the larger patches of IAP because they seem more threatening than smaller ones, because it produces more noticeable work, or simply because the impacts do not become obvious and problematic until the invaders are well-established and cover large areas (Moody and Mack 1988; Pyšek et al. 2020). Nevertheless, because small patches tend to expand more rapidly and cover a greater area (Cousens and Mortimer 1995), many authors have suggested that removing smaller patches before they reach considerable growth rates is crucial to halt the spreading in the future (Moody and Mack 1988; Campbell 1993). An additional reason to clear the invaded sites early on is that the intensity of the restoration intervention required increases with the invasion duration (Holmes et al. 2000). However, the patch size is not the only relevant criterion and not all the IAP clearings should be planned according to it alone.

Patch area becomes particularly relevant when dealing with species with long-distance dispersal mechanisms because regardless of how well the edge is contained, they can always spread long distances and start new foci. If, on the contrary, the species expands advancing gradually as a front, the edge length (interface with other LULC) is particularly relevant. In that case, the less edge the better, to halt the patch expansion. For such species, for which the perimeter is essential, but the area is still important, the Perimeter-Area Ratio (PAR) is appropriate to identify the patches to remove first. Using

PAR as the main criterion, the smaller patches are expected to be selected first since they usually display higher PAR values, but larger and irregular patches may also rank in top positions. In this regard, Minor & Gardner (2011) found that species with a high probability of random long-distance dispersal are best managed by focusing on the largest patches, while species more prone to short-distance dispersion are best managed considering the patch configuration.

This work aims at contributing to halt *A. dealbata* invasion in a Natura 2000 site in Central Portugal. For that, an analysis is conducted based on the species distribution and connectivity, leading to a patch prioritization list that informs which *A. dealbata* patches should be removed first to maximize the operation effectiveness.

4.1.2. Methods

4.1.2.1. Study area and species

The study area corresponds approximately to the Special Area of Conservation (SAC) "Serra da Lousã" (PTCON0060) in central Portugal (Figure 21). Lousã mountain's highest peaks range between 800 m and 1.200 m and display some very steep slopes and narrow valleys. The rough orography and influences by Atlantic and Mediterranean climates, contribute to diversified vegetation and make the site relevant from the landscape standpoint. The 15.157 ha of the SAC are almost entirely covered by forest (12.008 ha) and shrublands (2.423 ha). The forest is mainly formed by *Pinus pinaster* (58,5%) and *Eucalyptus globulus* (16,6%). The invasion by alien species is one of the major concerns, especially when considered together with the threat of wildfires, from which some of these species (e.g., *Acacia dealbata*, *Hakea sericea*) can capitalize to increase their distribution rapidly.
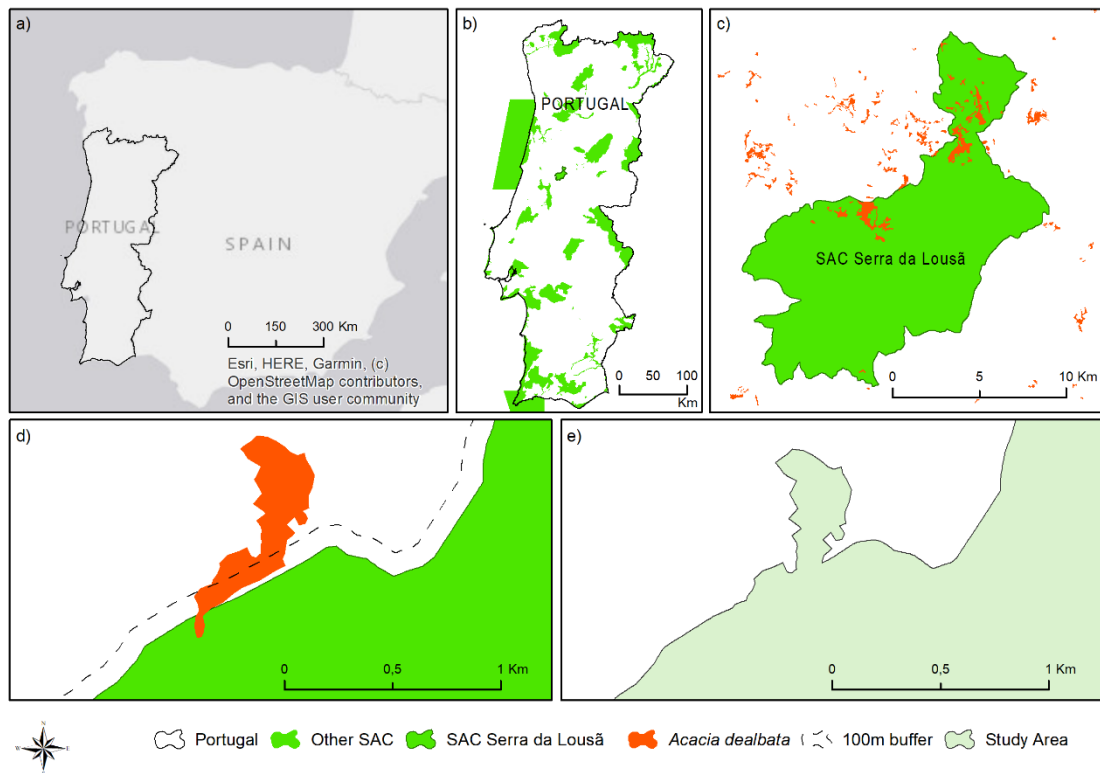
Figure 21 - Study area. a) Iberian Peninsula; b) Special Areas of Conservation (SAC) in mainland Portugal; c) SAC 'Serra da Lousã' and *Acacia dealbata*; d) detailed view of the 100m buffer; e) integration of the buffer and intersected *Acacia dealbata* patches in the study area.

Acacia dealbata

The silver wattle *Acacia dealbata* was first introduced in Portugal in 1850 for ornamental reasons and soil stabilization. Nowadays, the species, known for its high invasive potential, is present all over the country. The larger patches are primarily located in the North and Centre of Portugal, but it grows nation-wide along water stream banks and roadsides (Plantas invasoras em Portugal, 2020: *Acacia dealbata*. https://invasoras.pt/pt/planta-invasora/acacia-dealbata (accessed 21 May 2021)). Although the strips of IAPs in the stream banks and roadsides may cover small areas and appear harmless or least concerning, they have elongated shapes and consequently high PAR. They can also act as a reservoir of propagules that can be liberated in disturbance events (Parendes and Jones 2000) and are usually important vectors of invasion into protected areas (Landres et al. 1998).

Regarding dispersal processes and patterns, the *A. dealbata* seed bank is mainly located under the tree canopy and its density declines steeply away from canopy (Passos et al. 2017). The seeds are mainly ant-dispersed which limits their spreading distance (Gibson et al. 2011). Despite casual dispersal by humans (seeds inadvertently transported in clothes, tools, machinery, etc.) and abiotic dispersal by water, *A. dealbata* is more prone to advance as a front than to display frequent long-dispersal events.

4.1.2.2. Base maps, preliminary steps and scenarios

The invasive species patches were extracted from the official land cover map of Portugal of 2018 (COS 2018; Direção-Geral do Território 2019). Due to its detailed and heterogeneous land-cover classification scheme, quality, and reliability, this cartography is widely used to support the most relevant land planning and management-related policy procedures and scientific studies developed in mainland Portugal. The land cover category 'Invasive Species Forest' includes more than one species but *A. dealbata* is by far the most abundant invasive species (Global Biodiversity Information Facility; https://www.gbif.pt/). With a minimum map unit of 1 ha, it is safe to assume that most - if not all - the patches classified as 'Invasive Species Forest' in the study area represent *A. dealbata.*

The study area was defined based on the official Natura 2000 Network map, namely the SAC shapefile. We extracted the SAC "Serra da Lousã" (PTCON0060), applied an outwards 100m buffer, and included the *A. dealbata* patches intersected by the buffer (Figure 21). The decision to include the nearest patches outside the SAC is based on the premise that removing the invader within the SAC and leaving patches in the adjacent area would probably result in reinvasion in the short term (Landres et al. 1998). This way, we avoid overlooking the processes occurring in nearby territory that may be influencing the landscape patterns in the SAC (Pauchard et al. 2003).

The first task was to calculate the PAR for every *A. dealbata* patch in the SAC. Then, three intervention scenarios were defined (Figure 22). The first scenario involves removing the largest *A. dealbata* patch (area = 106,61 ha). The second scenario involves removing intermediary PAR patches that total an area as close as possible to the area of

the largest patch (n=5; total area = 107,49 ha). The third scenario involves removing the highest PAR patches up to an area as close as possible to the area of the largest patch (n=25; total area =98,94 ha).
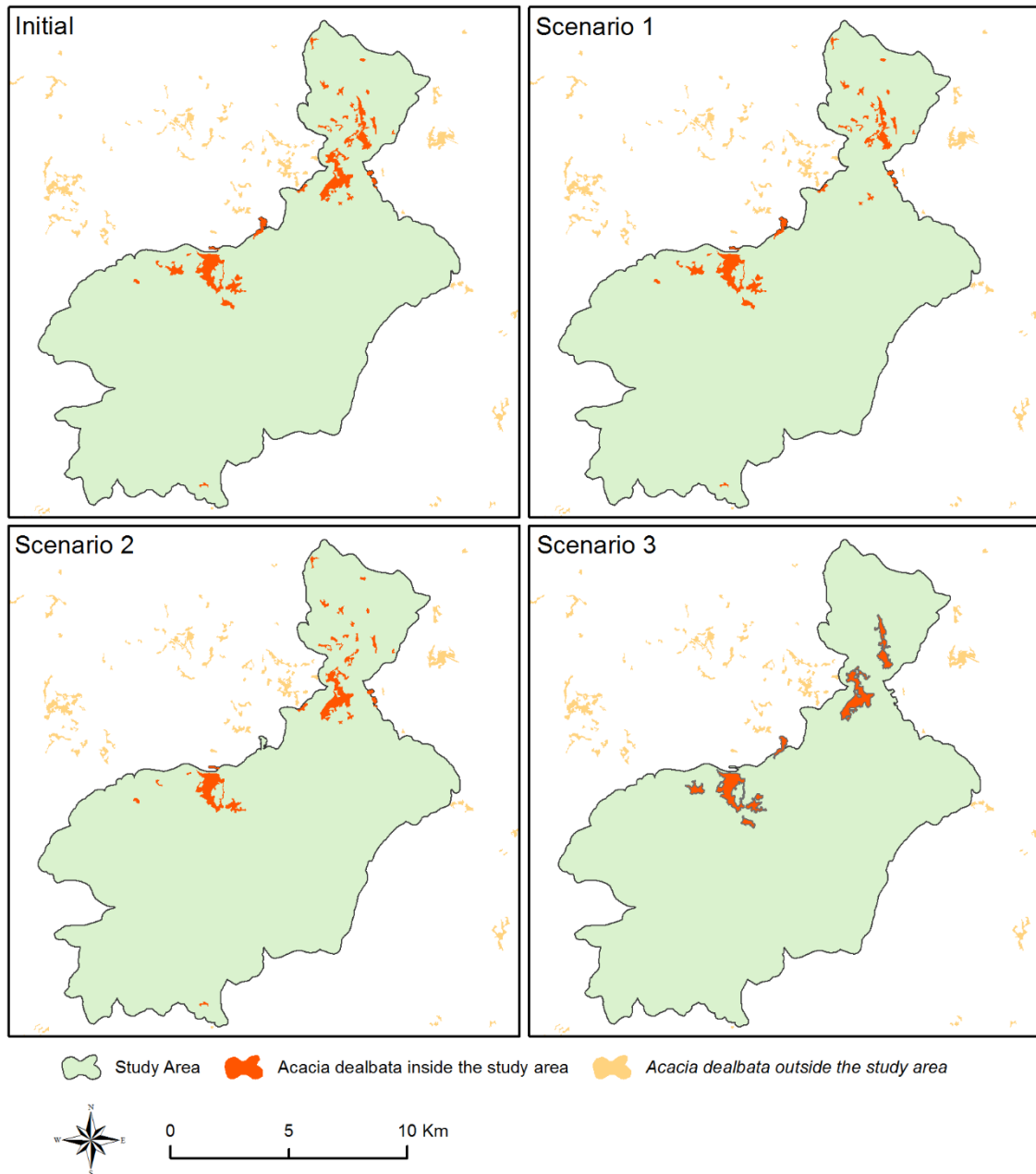
Figure 22 – Initial and scenarios' maps. In all scenarios the area of *A. dealbata* removed is similar. In scenario 1 the largest patch is removed. In scenario 2, five patches with intermediary perimeter-area ratio values are removed. In scenario 3, the 25 patches with higher perimeter-area ratio are removed.

4.1.2.3. Landscape dynamics analysis

To know what ToD the different interventions would produce we ran the simulations using LDTtool in ArcGIS 10.7. Because we did not know beforehand which spatial resolution would be appropriate, we ran the tool for each scenario using 500 x 500m, 1000 x 1000m, and 2000 x 2000m square grids.

4.1.2.4. Connectivity analysis

Once there is a set of patches identified for removal, several criteria can be used to rank them and help decide where to act first. A valuable assistance would be to know in advance how important each patch is for the species' connectivity in the study area. We assessed each patch's role to connectivity using the Conefor Sensinode software (Saura and Torné 2009) via the metric:

$$dM(\%) = 100.\frac{M - M_{after}}{M}$$

Where M is an overall connectivity metric when all patches are present in the landscape and M_{after} is the metric value after a determined patch is removed. Running the simulation for all *A. dealbata* patches, we obtain each patch contribution to the species connectivity in the study area. The higher the dM, the more important the patch is for the connectivity and the higher it should be ranked in the removal list. As overall connectivity metric (M), we used the numerator of the Integral Index of Connectivity (IIC_num) (Pascual-Hortal and Saura 2006) given by:

$$IICnum = \sum_{i=1}^{n}\sum_{j=1}^{n}\frac{a_i.a_j}{1 + nl_{ij}}$$

Where *n* is the total number of patches in the landscape, a_i and a_j are attributes of patches *i* and *j*, and nl_{ij} are the number of links between patches *i* and *j*. The threshold distance used was 100m and PAR was pointed as the patch key attribute.

4.1.3. Results

The analytical outcomes are i) basic instrumental data relative to each scenario (area, edge, and patches removed and remaining); ii) concrete results regarding landscape dynamics; iii) complementary results based on the connectivity assessment and iv) final list of patches to remove.

4.1.3.1. Scenarios

The existing situation and the resulting scenarios are spatially represented in the Figure 22 and the associated values (and variations) of area, edge and NP are present in the Table 4. For similar area amounts, the removal of many smaller patches instead of fewer larger ones causes the removal of a substantial extra edge length. For instance, removing the larger patch (scenario 1) subtracts 13.481m of edge, while removing the smaller patches totaling a similar area (scenario 3) leads to an edge decrement of 29.912m.

Table 4 – Area, edge length and number of patches in the initial situation and in the three scenarios. The values in parentheses are the differences between the scenario and the initial situation. Numbers in bold show that the edge length and Number of Patches (NP) are more affected in scenario 3.

	Initial	Scenario 1	Scenario 2	Scenario 3
Area (ha)	429,76	323,14 (-106,62)	322,26 (-107,49)	330,82 (-98,94)
Edge length (m)	75.760	62.279 (-13.481)	57.700 (-18.060)	**45.848 (-29.912)**
NP	33	32 (-1)	28 (-5)	**8 (-25)**

4.1.3.2. Landscape dynamics analysis

To ascertain which spatial resolution would fit the analysis better, all scenarios were run using 500 m, 1.000 m, and 2.000 m grids (Figure 23). The smallest square seemed appropriate because the minimum map unit is 1 ha and each square covers 25 ha, suggesting that most patches would fit in a single square, avoiding being counted more than once. However, the two largest patches are much larger than the majority and cross 15 and 12 squares. Using the 2.000 m squares would solve that problem but they are so large that important changes could go unnoticed. This grid size is usable, but there is no reason to select it if is possible to use a more detailed (analytical) scale without compromising the analysis. Based on these reasons, we decided to use the 1.000m grid.

Looking at the results obtained using the 1.000 m grid, in scenario 1 there is ToD H in few contiguous squares (n=6), in scenario 2 there are more H squares (n=13) and more spread across the study area, and finally, scenario 3 displays even a higher number of H squares (n=22) (Figure 24). Overall, for a similar area of intervention (~106,62 ha), the elimination of a large patch constitutes a more localized intervention. In contrast, the elimination of multiple smaller patches means a more spatially widespread intervention.
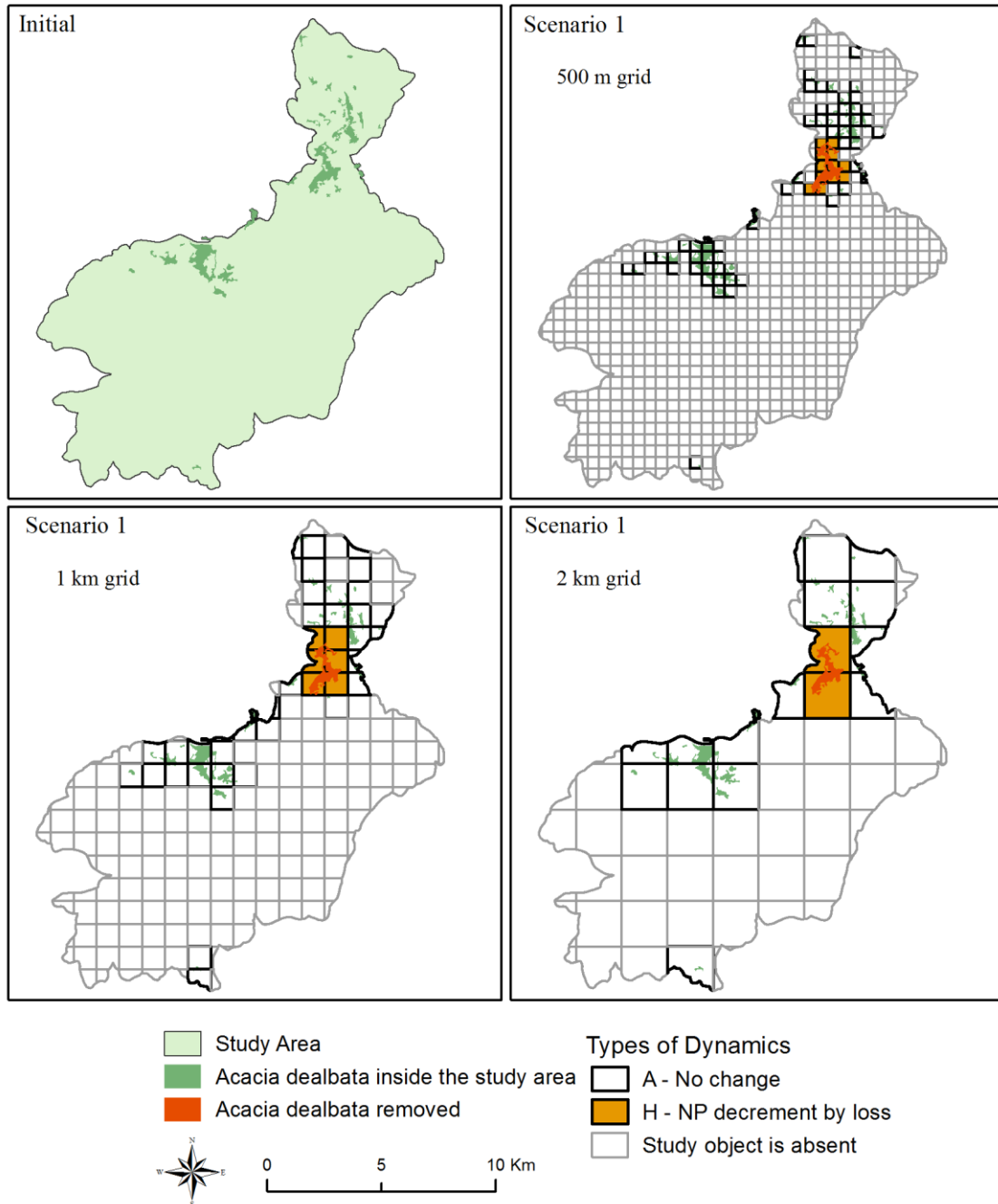
Figure 23 – Types of dynamics for scenario 1 calculated using grids with different sizes.
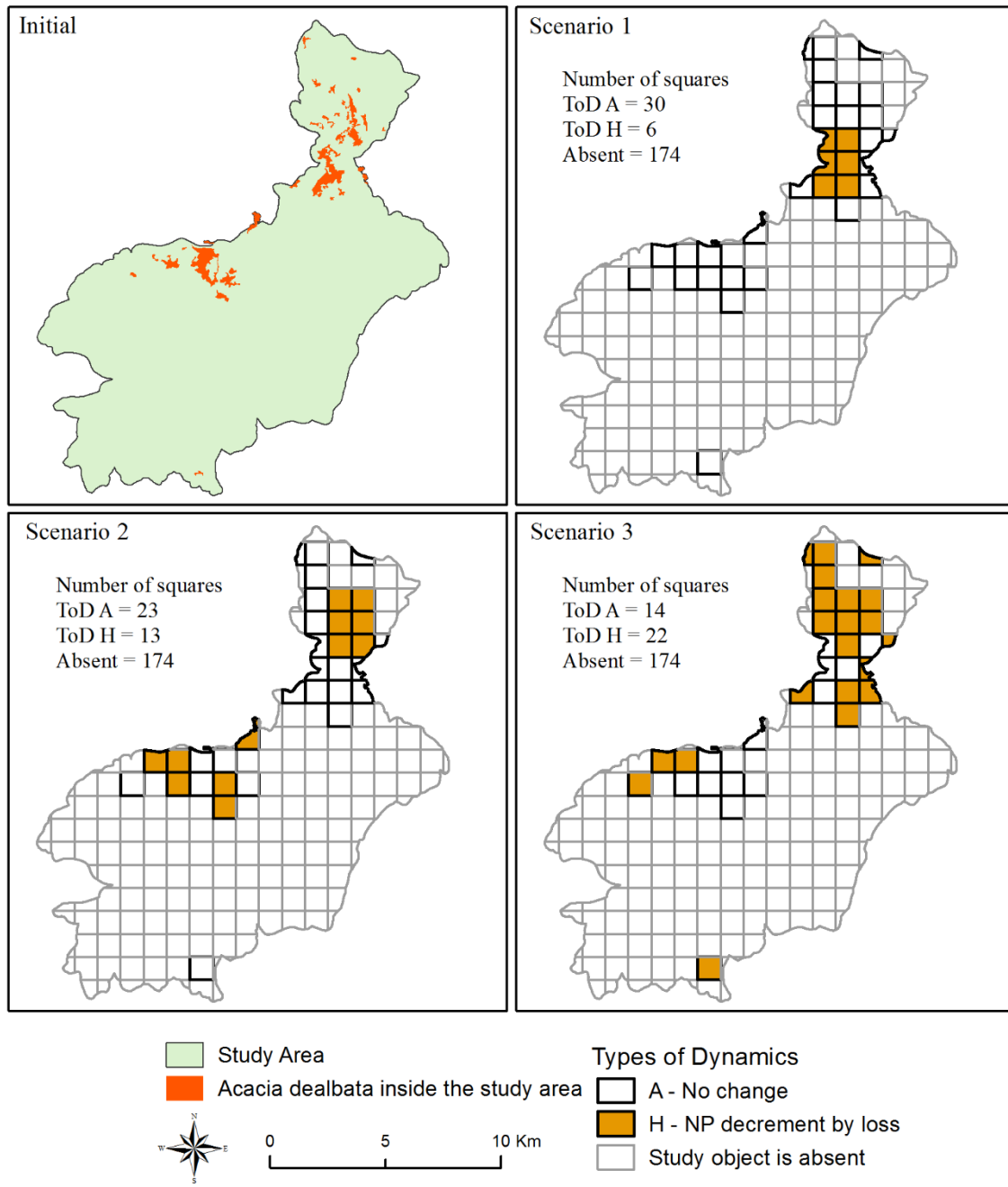
Figure 24 – Types of dynamics produced by the different scenarios.

4.1.3.3. Connectivity analysis

The connectivity analysis reinforces the spatial aspects and provides information about patch importance for the species connectivity. The output is a map where the *A. dealbata* patches are ranked according to how much they contribute to the species overall connectivity in the study area (Figure 25).
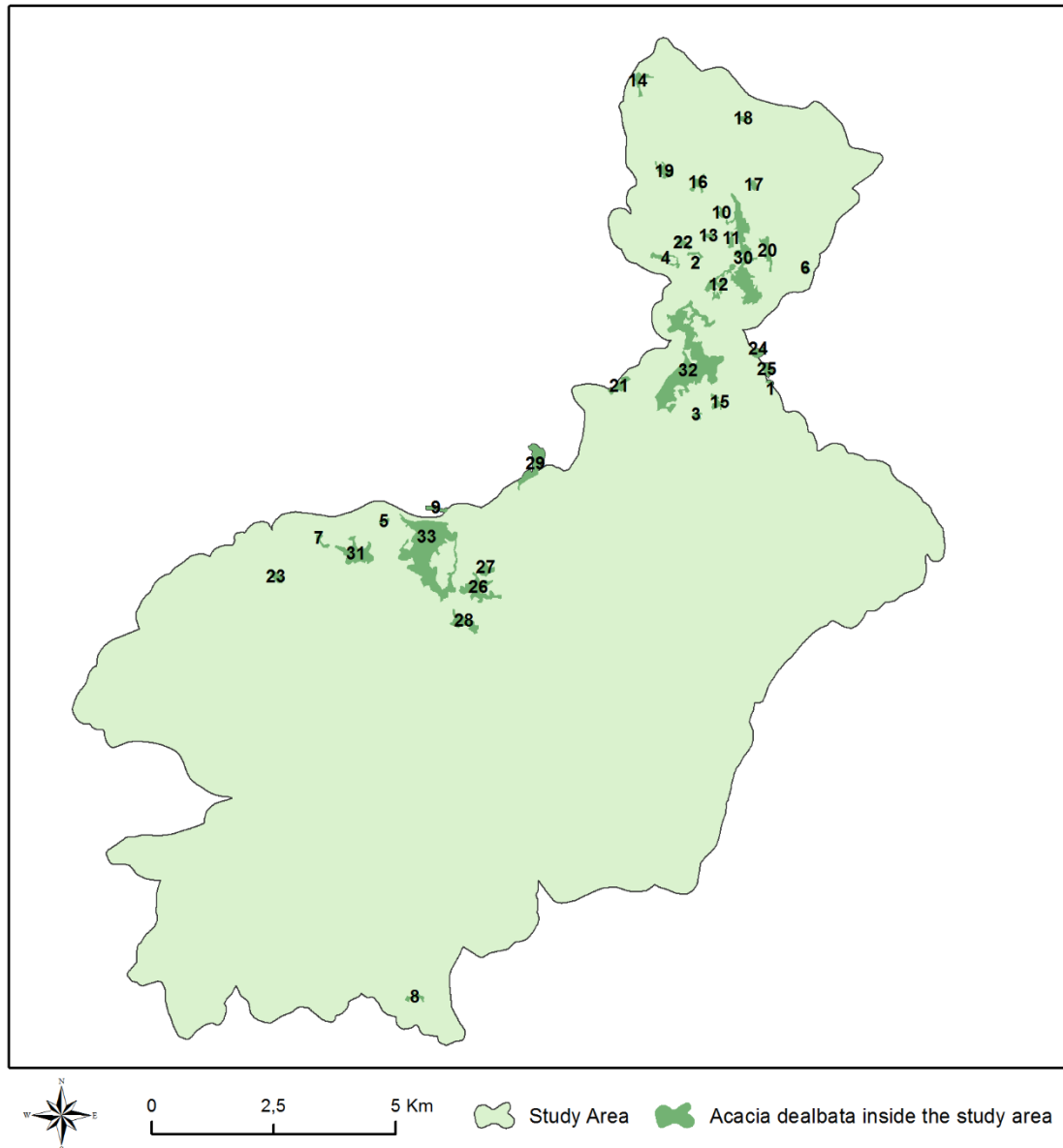
Figure 25 – Ranking of *A. dealbata* patches based on their relative importance for the species' overall connectivity in the study area. Calculated using perimeter-area ratio as the key attribute. From 1 – patch that contributes the most, to 33 – patch that contributes the least.

4.1.3.4. Combined analysis

The landscape dynamics analysis showed that scenario 3 produces more squares with 'ToD H – NP decrement by loss' by removing many smaller patches than other scenarios where fewer but larger patches would be eliminated. The connectivity analysis showed that the patches have different contributes, due to PAR and location, to the global species connectivity and may be ranked according to this criterion.

Combining both approaches, we propose a strategy to aim for scenario 3, which involves removing the higher PAR patches (mostly smaller patches), beginning with those with higher contribution to connectivity, as depicted in Figure 26.
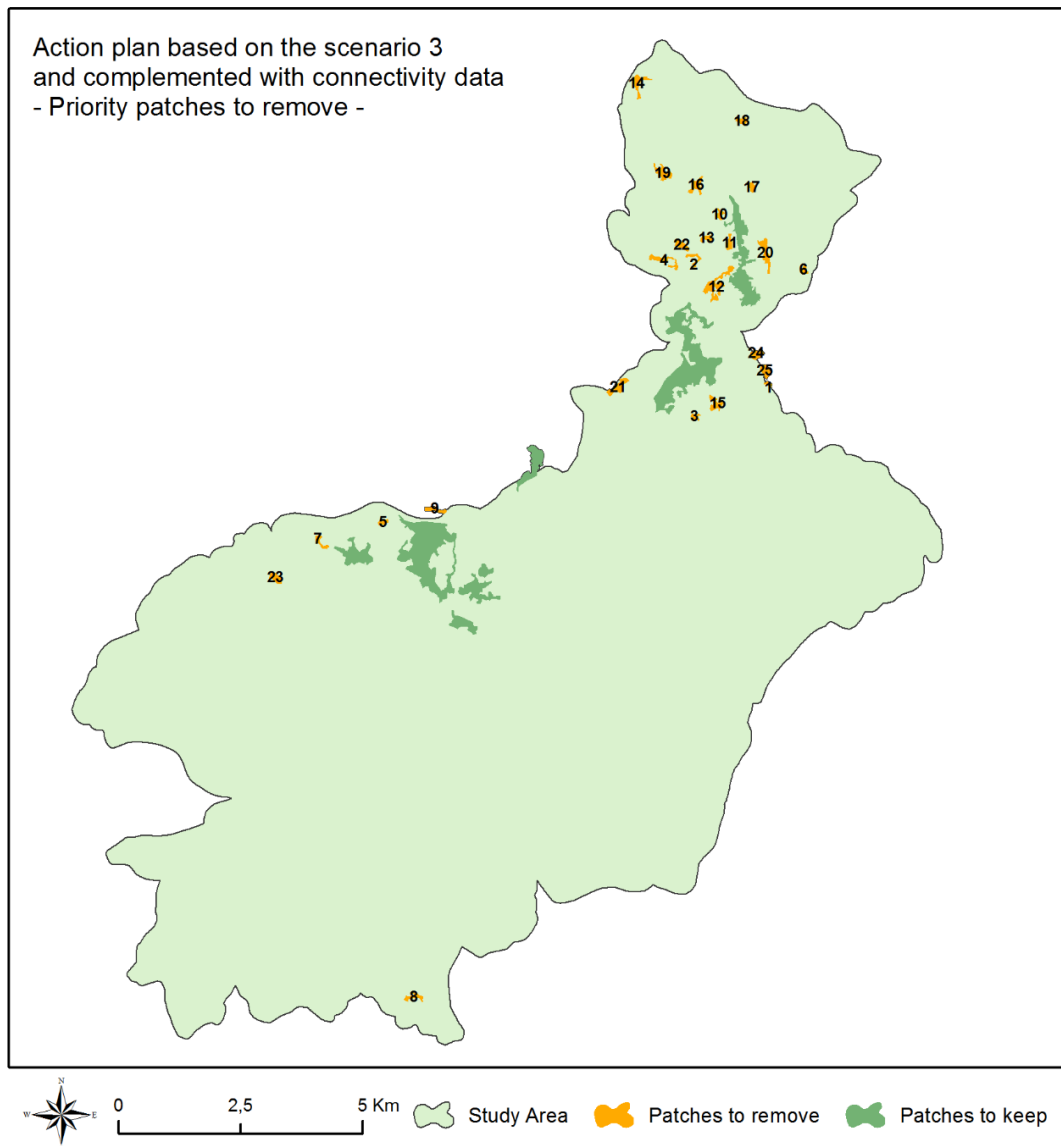
Figure 26 – Ranking of *A. dealbata* patches to remove towards scenario 3 and considering connectivity aspects.

4.1.4. Discussion

Restoring areas invaded by IAP is challenging and to increase the probability of success should be a key goal. In a IAP intervention program it is important to remove the more relevant patches as early as possible due to the difficulty to correctly estimate the required investment needed to succeed (Panetta 2009). But relevant according to what?

Removing one large patch may seem a good approach but removing many small patches may be more effective. In this case study, scenario 2 illustrates how an intermediate compromise could be a solution. Choosing scenario 2 over scenario 1 means removing additional 4.689 m of edge by subtracting four more patches. Implementing scenario 3 instead of scenario 2 would require the removal of 20 more patches to reduce the edge by an additional 11.852m. It is up to the manager to establish the cost-benefit threshold for the contexts at hand, and informative elements like these are helpful.

Given the biology of the *A. dealbata*, preventing the spreading is more efficiently achieved by eliminating the higher PAR patches (mostly smaller and younger satellite patches) due to their potential to boost the invasion rate. Therefore, one advantage of removing satellite patches first is that it prevents them from merging with other (satellite or parental) patches in the future (Pauchard et al. 2003). That situation, represented by ToD G – 'Aggregation by gain', provokes landscape homogenization and therefore should be avoided. Conversely, if invasion rate is more worrisome than homogenization, removing a patch that belongs to a cluster of patches, even if it highly contributes to connectivity, may not be a priority. In such case, the expansion potential would be naturally limited due to the proximity to other IAP patches, while isolated IAP patches could represent a more significant chance to quickly occupy adjacent territory. Such situation is present in the northern part of the study area where some patches are clustered and removing some of them can make sense to affect the IAP connectivity but do not necessarily represent the major impact in terms of halting the potential spreading.

According to Mack & Lonsdale (2002), ignoring small foci of IAP while focusing on major infestations, provides time for the once-inconspicuous satellite populations to

flourish. A clear example is *Schinus terebinthifolius*, which was introduced as an ornamental to South Florida but did not spread across the landscape until decades later (Ewel 1986). Ghersa et al. (2000), studying johnsongrass *Sorghum halepense,* found out that small foci uniformly distributed over a previously vacant area occupied that area more quickly than did the advancing front of an adjacent source population. Our proposal for dealing with *A. dealbata* in the SAC 'Serra da Lousã' follows the same line of thought of removing the higher PAR, and thus mostly the smaller patches. Among them, those potentially boosting 'ToD G – Aggregation by gain' should be eliminated first. Once the small foci are removed and the probability of rapid expansion is lowered, one can focus on the large patches.

Cadenasso & Pickett (2001) stated that a way to fight invasion expansion is to keep forest edges intact to function as a barrier to the flux of seeds while actively removing the established IAP on the edge. How deep the edge effect goes depends on several factors such as species, age, and density, but it is always influenced by the patch size and mainly by its shape (Laurance and Yensen 1991). In our case study, the IAP patches were almost surrounded by forest, mostly pine, and only one was completely surrounded by shrublands and thus supposedly facing less resistance to expansion. We had no further information about the stand age, density, edge composition or other variables that could be used as additional criterion to enrich the ranking process.

Our approach produced results that could be useful for an actual IAP management plan in the SAC, but it could be improved with more accurate and updated data. For instance, a detailed inventory of the populations is vital to avoid missing isolated patches or individuals that can jeopardize the subsequent analysis (Marchante et al. 2019). Martins et al. (2016), using remote sensing techniques, mapped *A. dealbata* patches smaller than 0,5 ha in a nearby location. More recently Ferreira et al. (2021) compiled a map using several sources, that includes a class of patches smaller than 0,1 ha. These works identified patches that we did not consider in our work. Moreover, ground surveys play a relevant role in the detection of new source and satellite populations (Radosevich et al. 2003) and thus, thorough fieldwork would be fundamental to obtain a more reliable representation of the situation in the terrain.

We suggest this approach for this species in this context and thus the analytical procedure here presented should not be seen as a 'one-size-fits-all' approach. We used

a spatially-explicit strategy focused on the satellite populations and incorporated connectivity information to enrich the analysis and support better decision-making. This method can be applied to large extensions of terrain but that may not be the best approach. A national scale ranking of patches is not necessarily more useful than the same type of information on a more workable scale, say regional or local. As Krug et al. (2010): "*The financial resources available determine the extent of the area which can be cleared, while the prioritization identifies the location of the areas to be cleared.*". For that reason, there is no need to prioritize an area much larger than the area that can be managed with the existing resources.

A useful use case we can anticipate for this method is the screening of large areas in early phases to support decision-making. For example, if a graph-based connectivity analysis identifies large *A. dealbata* components or 'connected regions' (groups of patches isolated from the other patches ((Pascual-Hortal and Saura 2006)) in the landscape, it could make sense to define one component as the study area, prioritize its patches according to the criteria found relevant and act to prevent 'ToD G – Aggregation by gain'. Weighing the pros and cons and considering the resources and constraints, the manager should be able to make an informed decision and adopt an appropriate strategy.

4.1.5. Conclusions

In a task as challenging and costly as restoring invaded areas, planners need to know where the resources can by employed more effectively. Based on the species dispersal traits, we argue that not only the area but also the perimeter and location of the patches should be considered when fighting the invasion. Addressing an *A. dealbata* invasion in a Natura 2000 site in central Portugal, three scenarios were designed and compared using the PAR, a landscape dynamics analysis, and a connectivity index. We conclude that removing the patches with higher PAR (mostly small satellite patches) would be more impactful than removing the larger patch or removing random intermediary PAR patches first. Further connectivity analysis provided an ordered list of patches to remove sequentially for higher effectiveness. As main implications for practice we highlight that

(i) removing larger patches of invasive plants is not always the most effective way to control the invasion, and (ii) patch size, shape and location all contribute to the distribution and dispersal of invasive plants and should be assessed to learn where a certain amount of work can produce optimal results.

Managers should adjust the prioritization strategies to the characteristics of each region (Roura-Pascual et al. 2010), but provided that the principles and main premises apply, this method seems viable for use in different geographic contexts for this species (and others with similar dispersal behavior). Since it is based on spatial metrics (NP, area and PAR) and their relations, the rationale is transferable to other similar contexts.

Overall, this approach can be valuable in the early steps of the planning process, supporting better decisions regarding the available resources and contributing to maximize the effectiveness of the action. In summary, our method for supporting the control planning of *A. dealbata* constitutes a strategy based on landscape dynamics fine-tuned with complementary connectivity information so that we end up with a clear, ordered list of patches to remove.

4.2. Grassland dynamics and preservation in Slovenia

This subchapter is based on the article:

Machado R, Santos P, Sousa-Neves N, Pirnat J. The recent spatiotemporal dynamics of grasslands in Slovenia: contribution to their preservation and management. (submitted)

4.2.1. Background

European seminatural grasslands were created by millennia of low-intensity and moderate land use (Hejcman et al. 2013). These systems have a high conservation value as they harbour diverse communities of animals and plants, some of which are unique and protected (Škornik et al. 2006; Veen et al. 2009; Dakskobler and Seliškar 2016; Jugovic et al. 2018; Dakskobler and Poldini 2019). The need to increase livestock breeding led to gradual deforestation, and over time more open landscapes with pastures were created. The adoption of more intensive agricultural practices, especially since the agricultural revolution of the 20th century, such as increased fertilization, mowing frequency and drainage, is associated with the deterioration of grasslands (Stoate et al. 2009; Van Vooren et al. 2018). Dry grasslands are also threatened by abandonment, mainly in marginal and less productive areas, which leads to overgrowth. As populations in rural areas age and young people move to cities seeking a different life, fewer people are able and willing to work the land (van Vliet et al. 2015). Extensive grasslands are an important source of public goods and services (recreation, tourism, carbon sequestration, quality meat and dairy products, wool, etc.). However, if these benefits remain unnoticed or unmonetized, people tend to avoid the laborious traditional ways of working the land.

Species-rich grasslands are relevant in Europe but have been rapidly decreasing (WallisDeVries et al. 2002) and are now amongst the most endangered ecosystems in the European Union (European Environment Agency 2020). Biodiversity preservation in agricultural lands is a major environmental challenge in Europe today, and thus, conservationists have been working on approaches to maintain the extensive use of grasslands and integrate them into modern and more profitable farming systems (Simoncini et al. 2019).

The intensity versus abandonment dichotomy is noticeable in Slovenia. Since 2000, a significant share of farms have abandoned livestock breeding or were abandoned altogether, while others increased in size and production intensity (Erjavec et al. 2018). This abandonment probably contributed to the recently observed deterioration and loss of farmland habitats.

Low-input extensive farming systems that support high levels of biodiversity can be considered High Nature Value (HNV) farmlands (Paracchini et al. 2008), which helps raise awareness of their importance. Grasslands have benefitted from several policies, from European to national, but the results have been far from expected. Kaligarič et al. (2019) labelled the results of grassland-specific agri-environmental measures (AEM) implemented via the European Agricultural Fund for Rural Development from 2007 to 2013 in Slovenia as an "*almost complete failure*".

Much of the grasslands' sustainability and capacity to host biodiversity depends on their actual use and management. In this regard, it is important to stress that land use is not necessarily equivalent to land cover because while the latter may identify a patch as grassland the former may provide essential information regarding the intensity, mowing frequency, etc. For instance, the grazing of small ruminants slows down the overgrowth process, but if too intensive, can impoverish the species composition (Dakskobler and Seliškar 2016).

Landscape structural indicators help understand biological processes and assess the habitats' overall quality or suitability for a given species. While generalists may thrive in heterogeneous landscapes (Devictor et al. 2008), specialists tend to respond negatively to the high heterogeneity of LULC (Assandri et al. 2019). Heterogeneity can reduce habitat availability/accessibility (due to fragmentation) and quality (due to disturbance). It is known that more specialized species tend to prefer large patches of grasslands and

are likely to suffer from intensified grassland use (Broyer et al. 2017; Tome et al. 2020). Šumrada, Kmecl, et al. (2021) recently conducted a study in Slovenia that shows that the highest diversity of grassland specialists was found in open landscapes with > 50% grassland and very low stocking density and concludes that the negative effects of increasing woody vegetation affect grassland specialists the most. Forest succession, especially when it occurs at a fast pace, is one of the critical drivers of farmland bird declines in Slovenia (Kmecl and Denac 2018). This process involves a gradual transformation of grasslands into forests that can occur with different spatial patterns in different locations and lead to distinct outcomes. For instance, losing a single grassland that acts as a stepping-stone that assures connectivity for several species can be worse than losing several closely clustered grasslands that provide redundant accessibility and therefore remain connected even if some disappear. In the same way, losing X ha of grasslands in a region where they are scarce can be worse than losing 2X ha in a grassland-dominated landscape.

Our concern is to ensure the coexistence of agriculture and biodiversity, and therefore we aim at producing useful information for grassland management and policymaking. We characterize the recent grasslands spatiotemporal dynamics in Slovenia, focusing on the amount and geometry, and discuss how the trends and forecasts relate to processes and concepts fundamental for biodiversity preservation, such as fragmentation, habitat availability and connectivity. Concretely, we expect to identify spatial patterns of grassland change, infer what the trends could lead to in the future, and point out specific actions or priorities (based on spatial dynamics) that can contribute to grasslands preservation, restoration, and improvement.

4.2.2. Methods

4.2.2.1. Overall characterization via landscape metrics

A brief characterization of grasslands' spatial patterns was made by calculating some landscape metrics using the Fragstats software (McGarigal et al. 2012). First, ArcGIS 10.4.1. (ESRI 2016) was used to merge the land cover categories of natural grasslands

and pastures, meadows and other permanent grasslands under agricultural use. The data was extracted from CLC maps for 2006, 2012 and 2018. The union of both categories was designated as 'pastures and natural grasslands' (PNG) and covers around 6.5% of the country. The CLC's minimum map unit is 25 ha, but clipping these FC within Slovenia's boundaries originates smaller polygons. We removed the polygons smaller than 0,1 ha (1.000 m^2) and used the remaining polygons in the analysis. We converted the three PNG maps to raster format (GeoTiff; 50 m pixel) and used them as inputs in FRAGSTATS with the parameters set as '8 cell neighborhood rule' and 'no sampling' strategy. The following class-level metrics were computed:

- Largest Patch Index (LPI)

LPI equals the percentage of the LULC category comprised by the largest patch, which makes it a measure of dominance. It is presented as a percent and approaches 0 when the largest patch of the LULC category of interest is increasingly small and reaches 100 if the entire landscape consists of a single patch of that LULC category.

$$LPI = \frac{\max{(a_{ij})}_{j=1}^{n}}{A}(100)$$

where:

a_{ij} = area (m^2) of patch ij.

A = total class area (m^2).

- Shape Index Distribution (SHAPE_MN)

The SHAPE_MN results from the computing Landscape Shape Index (LSI), that measures patch shape complexity, for each patch and calculate the mean. It equals the patch perimeter divided by the square root of the patch area and is adjusted by a constant for a square standard. It equals 1 when the patch is square and increases without limit as the shape becomes more irregular.

$$LSI = \frac{.25\sum_{k=1}^{m}e_{ik}^{*}}{\sqrt{A}}$$

where:

e_{ik}^* = total length (m) of edge in landscape between patch types (classes) i and k;

A = total landscape area (m²).

- Patch Cohesion Index (COHESION)

COHESION measures the physical connectedness of the corresponding patch type. The index quantifies connectivity in binary landscapes and ranges between 0 and 100 as it is calculated as a percentage. The value decreases as the proportion of the landscape comprised by the LULC category of interest decreases and becomes increasingly subdivided and thus less connected. Higher values reflect a more clumped or aggregated distribution.

$$COHESION = \left[1 - \frac{\sum_{j=1}^{n} p_{ij}^*}{\sum_{j=1}^{n} p_{ij}^* \sqrt{a_{ij}^*}}\right] \cdot \left[1 - \frac{1}{\sqrt{Z}}\right]^{-1} \cdot (100)$$

where:

p_{ij}^* = perimeter of patch ij in terms of number of cell surfaces.

a_{ij}^* = area of patch ij in terms of number of cells.

Z = total number of cells in the landscape.

- Effective Mesh Size (MESH)

MESH provides a relative measure of patch structure. It quantifies landscape fragmentation based on the probability that two randomly chosen points in a region will be connected (Jaeger 2000). MESH ranges between the ratio of cell size to landscape area and the total landscape area. The minimum value occurs when only a single pixel belongs to the LULC of interest and the maximum when all the landscape consists of a single patch of the LUCL of interest.

$$MESH = \frac{\sum_{j=1}^{n} a_{ij}^2}{A} \left(\frac{1}{10000}\right)$$

where:

a_{ij} = area (m²) of patch ij.

A = total landscape area (m²).

4.2.2.2. Landscape dynamic typology

The analysis was conducted via LDTtool and the first decision to make was which AU should be used to avoid the most possible the MAUP effects. An AU too large allows existing dynamics to go unnoticed as it calculates all the values and delivers only one ToD (e.g., loss for the whole square, instead of located losses and gains). An AU that is too small is even more problematic as it does not entirely contain many patches that will then be considered again in the adjacent AU. Thus, a small number of intersections (polygons partially outside the AU) indicates AU size suitability. For comparison, we tested squares of 1 km (100 ha), 2 km (400 ha), 5 km (2.500 ha) and 10 km (10.000 ha). The outcome of this preliminary screening, which is thoroughly presented in the '4.2.3.2 Landscape Dynamics Typology' (Results) section, allowed us to choose the 5 km squares to conduct the rest of the analysis.

The inputs and settings were the following: Tool – 'Landscape Dynamic Types 3M Squares'; Study Area Polygon – 'Slovenia'; Landscape Moment 1 – 'png_slo06' (pastures and natural grasslands); Landscape Moment 2 – 'png_slo12'; Landscape Moment 3 – 'png_slo18'; Keep patches equal or larger than – '1000 square meters'; Squares width and height – '5000 meters'; Output feature class – 'LDT_3M_Sq5km' (Figure 27).
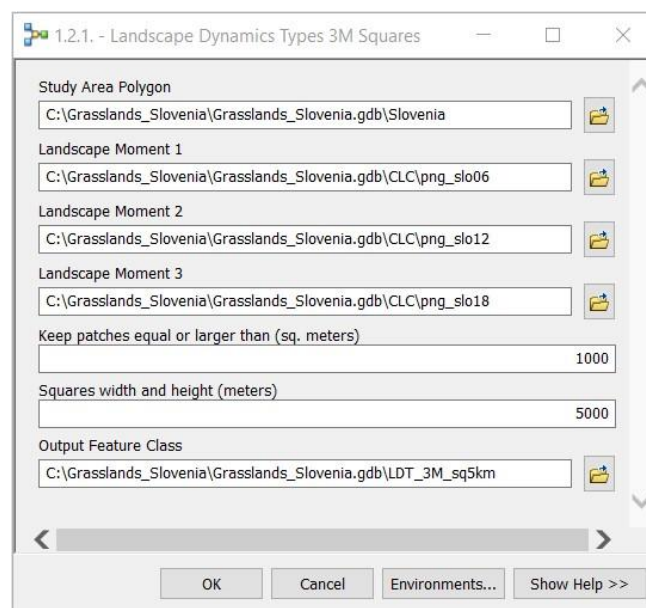
Figure 27 – Filled dialog box of the tool '1.2.1 – Landscape Dynamic Types 3M Squares'.

4.2.3. Results and Discussion

4.2.3.1. Overall characterization via landscape metrics

The results obtained in FRAGSTATS are shown in Table 5. The largest patch was the same in 2006 and 2012, so the LPI was equal, and the PNG was similar. In 2018, the largest patch was smaller and so was LPI. The largest patch represents 1,61% of the total PNG area in 2006 and 2012 and 1,29% in 2018, revealing that the largest patch is not dominant. The shape index did not vary much, meaning shape complexity was stable over time. This number concerns the average of the shape index for all PNG patches; therefore, some patches may be particularly irregular, but overall, the complexity was low, which matches the empirical experience of PNGs typically being regular patches.

The aggregation indices reveal no change or little increase from 2006 to 2012 and a more noticeable decrease from 2012 to 2018. COHESION and MESH assessed PNG fragmentation because they were computed at the class level. COHESION's advantage is that it is displayed as percentages, making it easy to understand when looking at the values of around 95–96% where PNG areas are mostly aggregated. This means that most of the PNG 50x50 m pixels (n = 543.368 in 2006; n = 543.402 in 2012; n = 506.817 in 2018) are physically connected. Notice that configuration is assessed here, not composition. If there was a single patch of PNG, regardless of its size, COHESION would be 100 because all of the existing area would be connected. On the contrary, if two or more patches were comprising a larger area, COHESION would be lower because not all of the PNG areas would be merged into a single patch. MESH brings a more functional character to the analysis. From 2006 or 2012 to 2018, its reduction reflects a decrease in the probability that two randomly chosen points in a region are connected. In practice, this means that in 2018 it was less probable that two individuals in two different locations (within PNG) had access to one another. This suggests that the recent dynamics in these systems may be harmful to the species that highly depend on them. In brief, the PNGs in Slovenia are mostly aggregated, although not dominated by a single large patch, and formed mainly by patches with low complexity (regular edges). The recent trend of area loss will likely harm the dependent species by diminishing the availability of and accessibility to these areas.

Table 5 – Landscape metrics values of Pastures and Natural Grasslands in 2006, 2012 and 2018. LPI – Largest Patch Index; SHAPE_MN – Shape Index (mean value); COHESION – Patch Cohesion Index; MESH – Effective Mesh Size.

	LPI	SHAPE_MN	COHESION	MESH
PNG_06	1,61	2,23	95,63	228,24
PNG_12	1,61	2,26	95,63	228,35
PNG_18	1,29	2,22	95,51	196,34

4.2.3.2. Landscape Dynamics Typology

The 2006–2012 period shows little change in the grasslands across Slovenia (39 patches were lost and there was a 10 ha increase). In the period of 2012–2018, PNG lost 65 patches and 9.249 ha, which is substantially higher. The trend in the second period overwhelms the first period's relative stability and establishes a global change between 2006 and 2018 of loss of 104 patches and 9.239 ha (Table 6).

Table 6 – Area and number of patches of Pastures and Natural Grasslands in 2006, 2012 and 2018. Largest patch and Number of Patches overcounts to assess the adequate square sizes.

	2006	2012	2018
Area (ha)	135.929	135.939	126.690
Number of patches (NP)	1.877	1.838	1.773
Largest Patch (ha)	1.567	1.567	1.475
NP using 1 Km Square (100 ha)	9.829	9.787	9.209
NP overcount	5,24	5,32	5,19
NP using 2 Km Square (400 ha)	5.568	5.528	5.235
NP overcount	2,97	3,01	2,95
NP using 5 Km Square (2500 ha)	3.272	3.232	3.074
NP overcount	1,74	1,76	1,73
NP using 10 Km Square (10000 ha)	2.557	2.518	2.400
NP overcount	1,36	1,37	1,35

In the LDT's framework a lesser amount and fewer patches suggests the occurrence of a ToD 'H - NP decrease due to loss'. Two indicators are important for determining which square size is more suitable for the analysis: the largest grassland patch and the NP overcount (given by NP count/actual NP). The largest patch of the three maps was 1.877 ha. Since the patch size displays a left-skewed distribution, the existence of a few patches larger than the AU does not necessarily jeopardizes the analysis. LDTtool was run four times using 1 km, 2 km, 5 km and 10 km squares. A 1 km x 1 km square covers 100 ha and the 2 km x 2 km covers 400 ha, and thus should only be used if no better alternative is available. A larger AU means fewer patches overlap squares and, therefore, a lower value of NP overcount. A balanced selection must consider both indicators and never lose sight of the goal of extracting valuable information. After discarding 1 km and 2 km squares, the selection was between the 5 km and the 10 km squares. Although the latter displays a lower NP overcount, there is not much difference between them (~1,36 vs. ~1,75) (Table 6).

Concerning the analytical scale, the 5 km square with its 2.500 ha seems appropriate to assess a dataset composed mostly of small patches with the largest patch covering 1.877 ha. On the contrary, much activity could happen and remain unnoticed inside a 10 km square with 10.000 ha. Based on this rationale, we decided to conduct the remaining analytical procedures using 5 km squares, which resulted in the outputs presented in Figure 28 and Table 7.
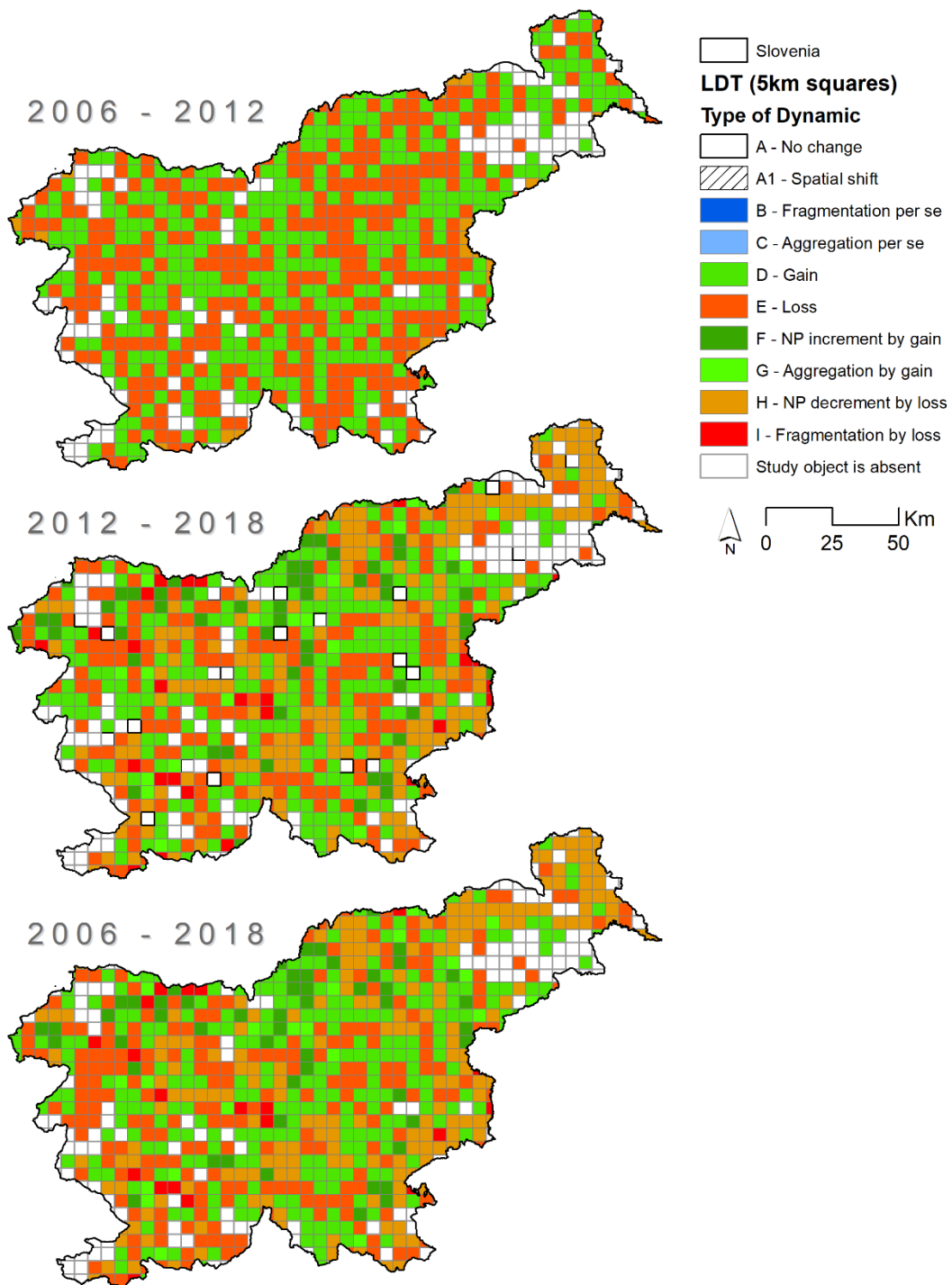
Figure 28 – Landscape Dynamic Types maps of Pastures and Natural Grasslands in Slovenia
(periods 2006-2012, 2012-2018 and 2006-2018).

Table 7 – Landscape Dynamics Typology results for the three periods analyzed. Number of squares assigned to each type of dynamic.

Type of Dynamic	2006-2012	2012-2018	2006-2018
A – No change	0	21	0
A1 – Spatial Shift	0	0	0
B – Fragmentation *per se*	0	0	0
C – Aggregation *per se*	0	0	0
D – Gain	382	245	258
E – Loss	367	206	223
F – NP increment by gain	0	62	54
G – Aggregation by gain	0	22	23
H – NP decrement by loss	24	190	197
I – Fragmentation by loss	0	27	20
Absent	158	158	156
Gain related	382	329	335
Loss related	391	423	440
Aggregation related	24	212	220
Fragmentation related	0	89	74

All analytical periods lack strictly geometric patterns (ToD A1 – Spatial shift, B – Fragmentation *per se* and C – Aggregation *per se)*. There are more squares with ToD related to loss (E, H and I) than those related to gain (D, F and G) in all periods, as well as a predominance of squares related to NP reduction (C, G and H) compared to those related to fragmentation (B, F and I). That is consistent with the overall trend at the national scale and suggests a PNG NP decrement due to area loss between 2006 and 2018.

In a very simplistic way, we can say that species benefit from habitat gain and suffer from habitat loss. The same cannot be said about fragmentation and aggregation *per se* because they can affect different species in different ways (Fahrig 2003). Specialists usually suffer from fragmentation (more heterogeneity, more edge effect), while generalists do not suffer and may even thrive with reasonable levels of fragmentation. If there is enough habitat and a species benefits from edge effects, a loss of habitat could be good if it breaks a patch into two or more. Provided that the remaining patches are

still accessible, their perimeter is now longer and the edge effect higher. Fragmentation can pose challenges to conservation because more patches need to be preserved to ensure a certain area is maintained, making patch prioritization more difficult (Pereira 2018). At the national scale, PNG lost area and patches, which is likely to harm the species that rely the most on them. Generalists with good mobility may be an exception. Zooming in to the AU scale, specialists are expected to be positively affected in the squares with PNG gain and negatively affected in those with PNG loss. Again, such a direct conclusion cannot be drawn regarding generalists. It depends on the magnitude of the changes and, ultimately, on how much they need PNG to fulfill their needs. Because this study focuses on the landscape changes themselves and not on a particular species or group of species, this is how far we can go in linking PNG dynamics with concrete impacts on fauna. Species-focused studies would require the use of parameters such as home range and distance dispersal threshold to calculate habitat availability and connectivity for the species of interest.

Additional information can be extracted from the outputs by projecting the current trend into the future. This is not a forecast as there are multiple drivers of landscape change, but it still provides insights on which direction the PNG can evolve based on the current trends. According to the most recent trend (2012–2018), 27 squares assigned with 'ToD I – Fragmentation by loss' and 206 squares assigned with 'ToD E – Loss' may be on their way to becoming 'ToD H – NP decrement by loss'. That is the final stage before a land cover category disappears in that square (Figure 5). A similar rationale could be made for the squares that registered PNG gain in the same period, but the conclusions are not equivalent for the reasons presented next. The comparison would be that the squares with 'ToD D – Gain' and 'ToD F – NP increment by gain' could keep gaining PNG until they reach the 'ToD G – Aggregation by gain'. This is conceptually correct but not likely to happen because pastures maintenance require active management but to lose pastures it is enough to abandon them. Unless natural succession works in our favor, it is easier to lose than gain the LULC we want because inaction leads to loss, and gain is only obtained through intentional action.It may be hard to maintain pastures in their current state, but it would be even harder to form new pastures or recover them once afforestation occurs. Besides that, in Central European grasslands, historical landscape structure seems to be an important predictor of current

species richness across different taxa and functional groups (Scherreiks et al. 2022). Thus, ecosystem restoration does not imply the immediate recovery of communities.

4.2.4. Conclusions

In Slovenia, PNGs are composed of several patches that display simple shapes and are not dominated by a single large one. Their recent spatial dynamics vary nationwide in direction (loss or gain) and magnitude, but the dominant trend shows area loss and NP decreasing (ToD H – NP decrement by loss). Such changes directly impact the more specialized and/or less mobile species (e.g., invertebrates, small birds, etc.) that are gradually being deprived of these biotopes.

Kaligarič et al. (2019) warned that the misclassification of grasslands prevents the correct application of compensation payments to HNV farms as often rewards others that are not. Incorrect classification and poor monitoring contribute strongly to the failure of the policies and thus need to be addressed and rectified urgently. An alternative (or complementary) approach are result-based schemes (RBS), in which the outcomes, not just the measures themselves, are the success criteria to unlock payments. Studies conducted in Slovenia show that grassland preservation would benefit from a RBS (Šumrada et al. 2021b) and that it would be well accepted by many farmers that prefer this approach over the management-based scheme, both in terms of payment and monitoring (Šumrada et al. 2022). Whichever funding scheme is in place, our results advise halting generalized PNG loss as the highest priority. Therefore, PNG maintenance and improvement can benefit from policies and concrete measures that:

- Prevent the disappearance of small patches and, if possible, increase their area. The smaller the patch, the more prone it is to disappear due to outside (matrix) influences or stochastic events. In this framework, relevant patches would be the smaller ones located in squares with area loss (ToD E – Loss, ToD H – NP decrement by loss and ToD I – Fragmentation by loss).
- Focus on patches that may be stepping-stones keeping other PNGs connected. This requires functional connectivity calculations with parameters derived from the characteristics of the species of interest.

- Allow or incentivize the conversion of other LULC into PNGs, ideally promoting the fusion of initially separated patches (promoting ToD G – Aggregation by gain and increasing the mean patch size). For increased probability of success, this should be attempted where previous PNGs were formerly established. Recently abandoned PNGs with overgrowth and earlier stages of afforestation could be appropriate lands for PNG restoration.

We provide general recommendations focused on PNGs as a system that needs intervention to be maintained and improved. Overall, such recommendations are expected to benefit the specialist species that depend on these systems. Nevertheless, when the goal is to preserve a certain species, its traits and requirements need to be addressed and considered. For example, species with low mobility (e.g., invertebrates) or birds such as *Alauda arvensis*, *Anthus trivialis* and *Saxicola rubetra* that require large areas of extensive meadows, and whose numbers show a worrisome decline (Kmecl et al. 2020), are greatly affected when their habitat becomes scarcer and divided. Large PNG patches are also important because they are probably more resilient and likely to harbor more grassland species than smaller PNG patches. For that reason, it is essential to prevent area loss, especially in spatial patterns that imply fragmentation (ToD I – Fragmentation by loss).

As a closing remark, we reinforce that PNGs are important for their social and cultural history, their ecosystem services, and their well-known biodiversity value. By identifying and locating the different spatial patterns of recent PNG dynamics, our work provides actionable knowledge to design policies to preserve this land-use system.

4.3. Spatial patterns in *Montado* loss

This subchapter contains elements of the article published in *Landscape Ecology* as:

Authors' contributions:
Rui Machado: Conceptualization, Methods, Analysis, Writing the original draft; Sérgio Godinho: Map production, Methods, Analysis; Nuno Guiomar: Writing; Artur Gil: Review; Janez Pirnat: Review and Supervising.

The *montado* as study object, the study area and the materials (*montado* maps) remain, but the subsequent analysis differs. The analysis presented in the published paper concerns connectivity while the analysis described in this subchapter, and still in preparation for submission, focuses on the loss of *montado* patches, particularly the smaller ones.

4.3.1. Background

The Portuguese *montado* is a multifunctional agro-silvopastoral system, similar to the Spanish *dehesa*, that covers most of the Southern region of the country, Alentejo (Pinto-Correia et al. 2011b). It is a complex system shaped by site variability (soil, climate, topography, etc.) and influenced by multiple production activities (agriculture, livestock, etc.) that share the same space in the landscape (Pinto-Correia 1993). The system has been sustainable for centuries based on a mixture of core products such as cork, cereal and livestock, complemented by others like honey, wool, firewood and charcoal, etc. (Gaspar et al. 2007). The optimized exploitation of the three pillars, agriculture, forest and livestock, requires a balanced management associated with different vegetation layers that fosters habitat heterogeneity making the landscape more prone to harbor a greater species diversity (Rosenweig 1995; Bugalho et al. 2011; Godinho and Rabaça 2011). Due to its role in harboring biodiversity, *montado* is recognised by the European

Environmental Agency as High Nature Value Farming system (Paracchini et al. 2008; Pinto-Correia et al. 2018).

Despite its importance at different levels, *montado* has been declining in area and quality, and its sustainability in the long-term is threatened (Godinho et al. 2016c). In a way, *montado* faces similar threats to those mentioned in the previous subchapter concerning the grasslands in Slovenia; system simplification provoked by land use intensification in the more productive areas and abandonment of the marginal ones (Pinto-Correia and Mascarenhas 1999). Evergreen canopy cover has been decreasing rapidly in the recent decades in mainland Portugal, especially in holm oak woodlands. This is a worrisome indicator, since canopy loss can be interpreted as a proxy (or precursor) of forest degradation. The reasons for this declining trend of *montado* are several, from mismanagement to global change factors such as higher mean temperature or the emergence of oak diseases (Acácio et al. 2021). The fruit dispersal of the two dominant tree species, *Quercus suber* and *Quercus rotundifolia*, occurs through barochory and synzoochory mainly by the European jay (*Garrulus glandarius*) and the wood mice (*Apodemus sylvaticus*) (Gómez 2003; Muñoz and Bonal 2007), but the low seedling survival is a constraint to tree recruitment (Leiva and Fernández-Alés 2003; Pulido and Díaz 2005). Overgrazing and use of heavy machinery are also pointed out as factors for the generalized lack of tree regeneration (Dinis et al. 2015; Almeida et al. 2016). The accumulation and interaction of ecological and anthropic disturbances over a long period have affected significantly the regenerative capacity of Mediterranean ecosystems (Blondel 2006). Poor recruitment and tree aging must be reversed to guarantee the maintenance of these valuable landscapes (Plieninger et al. 2010) and the more degraded the system, the less prone it is to recover, due to establishment of negative feedback loops. For instance, *montado* loss leading to fragmentation, and thus decreasing the connectivity for some animal species, including its seed dispersers, hampers seed dispersion and overall expansion (Herrera and García 2010; Puerta-Piñero et al. 2012).

Not all *montado* is the same as the system can vary a lot in its structure and design - tree species, age and density, livestock species and amount, dominance of shrublands or pastures, etc. (Pinto-Correia et al. 2011a). Despite the type of *montado*, it is expected that large continuous patches are more resistant to changes, and resilient to

disturbances, than smaller scattered ones. We hypothesize that small patches have been disappearing more than large ones, which empirically seems evident, but requires further investigation. The main goal is to gather information concerning the lost patches and their dimensions, look for patterns and, based on the outcome, be able to scenarize into the future. That kind of knowledge is particularly valuable and actionable since *montado* is a human-made and maintained system, highly dependent on management decisions. That is to say that *montado* patch dynamics is not only subject to natural processes but also and very deeply dependent on human decisions, that should be taken with the best and more comprehensive information available.

4.3.2. Methods

4.3.2.1. Study area and montado maps production

The study area was selected according to the biogeography boundaries and comprises the Alto Alentejano Superdistrict (Costa et al. 1998). The area covers 856.720 ha in a region where *montado* is a typical land use system (Bugalho et al. 2009) and displays its predominance by occupying about 45% of the area, followed by arable land that covers about 28%. In this region, the *montado* is mostly formed by varying densities of cork oaks (*Quercus suber*) and/or holm oaks (*Quercus rotundifolia*) (Godinho et al. 2016a) with natural pastures or cultivated understory (Canteiro et al. 2011). The climate is markedly Mediterranean with hot and dry summers and wet and cold winters. The mean annual precipitation ranges between 550 mm and 650 mm. The relief is predominantly soft and elevation ranges from 40 m to 645 m.

Figure 29 – Study area.

Montado is a complex land use system hard to equate to a specific LULC category. It can exhibit so much internal variability at multiple dimensions that it is hard to define it concisely and map it. The tree density variability and fuzzy boundaries, in particular, make the *montado* map production very challenging (Van Doorn and Pinto-Correia 2007) but some works have shown that it is possible to obtain fairly accurate maps via satellite imagery and machine learning algorithms (Godinho et al. 2016a, b; Allen et al. 2018).

In this study, *montado* maps for the years 1984, 1999 and 2014 were produced using satellite imagery from Landsat Thematic Mapper, Enhanced Thematic Mapper Plus and Operational Land Imager sensors (path 203 and row 33). Spring and summer scenes were acquired to optimize inter-class separability based on the seasonal vegetation variation (Rodriguez-Galiano and Chica-Olmo 2012; Godinho et al. 2016b). Adjustments were made to tackle the absence of cloud-free images in the springs of 1984 and 1999 (images from 1985 and 2000 were used instead). After atmospheric correction was applied using the FLAASH (Fast Line-of-sight Atmospheric Analysis of Spectral Hypercubes) method, six vegetation indices were calculated: Enhanced Vegetation Index (EVI), Short-Wave Infrared Ratio (SWIR32), Carotenoid Reflectance Index 1 (CRI1), Green Chlorophyll Index (CIgreen), Normalised Multi-band Drought Index (NMDI), and

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Soil-Adjusted Total Vegetation Index (SATVI). More information about the calculation and effectiveness in semi-arid environments are available in Godinho et al. (2016b).

The classification was conducted using the Stochastic Gradient Boosting (SGB) algorithm (Friedman 2002) based on 1.300, 1.301, and 1.549 sample points (80% for training and 20% for validation) for the years 1984, 1999, and 2014, respectively. The points were collected through Landsat image-interpretation, following a stratified-sample approach based on eleven land cover types: (1) *montado*, (2) pine forest, (3) eucalyptus forest, (4) olive groves, (5) vineyards, (6) irrigation agriculture, (7) dry crops/pastures, (8) shrublands, (9) water bodies, (10) bare soils and (11) urban areas (Table 8).

Table 8 – List of land cover types and number of sample points used for classification.

Class Name	Number of sample points		
	1984	*1999*	*2014*
Montado	417	417	420
Eucalyptus Forest	80	83	117
Shrubland	80	81	81
Pine Forest	80	80	80
Water	90	85	89
Olive Grove	110	102	148
Irrigation Agriculture	80	81	101
Dry crops/pastures	123	117	213
Bare soil	80	80	81
Urban	80	80	80
Vineyards	80	95	139

The following cross-validation was performed using auxiliary geoinformation: the Portuguese official land-use map (Carta de Uso e Ocupação do Solo) of 1990 (Direção-Geral do Território 2019) for the 1984 map; high-resolution true-color orthophotomaps, dated 2005 (produced by CNIG - National Center of Geographic Information), for the 1999 map; and high-resolution imagery from 2011-2013 available in Google Earth, for the 2014 map. The SGB classification produced moderate agreement and good accuracy: 1984 (Overall classification accuracy (OA) = 81,85%; Kappa coefficient (K) = 0,78), 1999 (OA = 75,58%; K = 0,71), and 2014 (OA = 80,07%; K = 0,77). Regarding the *montado* areas, the classification also produced reasonable accuracies: 1984 (Producer's Accuracy (PA)

= 89,2%; User's Accuracy (UA) = 74,7%), 1999 (PA = 91,6%; UA = 80,0%), and 2014 (PA = 85,7%; UA = 75,8%).

4.3.2.2. *Montado* dynamics

The LDTtool was used to calculate the ToD for the three analytical moments using 1 km x 1 km squares. This square size was chosen because we knew beforehand that the subsequent analysis would focus on small patches. Since the smallest mapped patches cover 1 ha, a small AU was required to detect the influence of such patches on the ToD. With a large AU (e.g., 5 km x 5 km or larger) the variation of small patches would be diluted in the global changes and remain unnoticed.

The largest lost patches covered 127 ha and 113 ha, in the first and second period, respectively. We built 5ha-range intervals (size classes), with the maximum value of 130 ha to encompass all the lost patches. Exceptionally, the first interval does not display a range of 5 ha due to the 1 ha minimum map unit.

Although the lost patches were the central topic of the study, their number had to be compared to the existing patches, for reference. Therefore, the number of existing *montado* patches in 1984 and 1999 as well as the accumulated area were calculated for each size class. Chi-square tests were applied to the existing and lost patches datasets to investigate whether the patches of the different size classes have been disappearing according to their availability or, on the contrary, some size classes lost more, or less, patches than expected. Next, via a few geoprocessing operations (Figure 30), the information regarding the ToD that had occurred in the corresponding square(s) was attached to each lost patch.
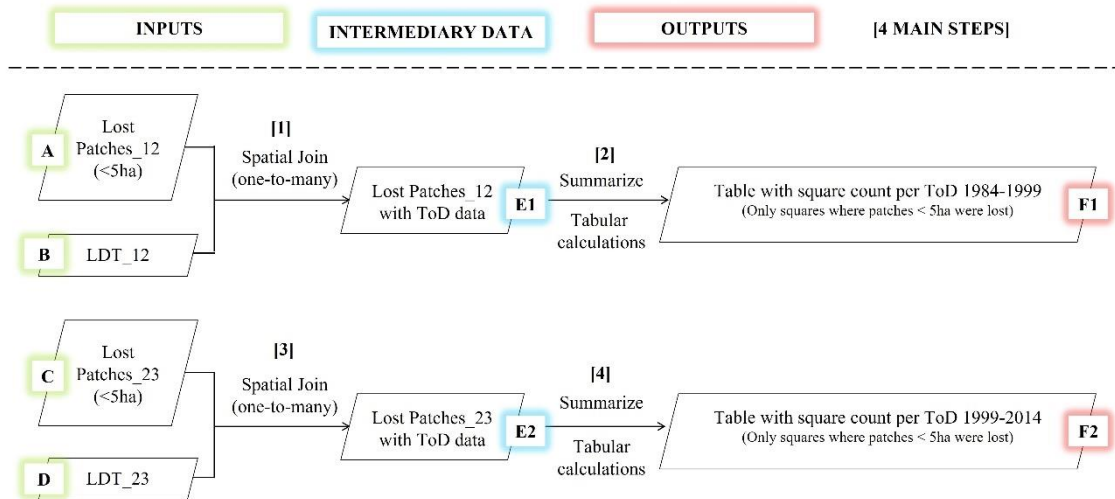
Figure 30 – Steps to combine types of dynamic and lost patch data.

4.3.2.3. Patch loss simulation

In the 1984-1999 period, 49% of the *montado* patches smaller than 5 ha were lost. In the period 1999-2014, the number was 59% which shows an acceleration compared to the earlier period. Based on these numbers, we built a conservative scenario where 50% of the *montado* patches smaller than 5 ha existent in 2014 would disappear next. In 2014, there were 3.868 patches in that size class, so 1.934 (50%) were randomly selected and deleted. This was actually done three times to provide us with more analytical data volume and diversity and originated three scenarios that differ in the eliminated patches that were randomly selected (Figure 31). The spatial distribution of the lost patches (smaller than 5 ha) in the previous periods was scattered, not clustered, in any particular region, which allowed us to use a random selection with no additional specific criteria.
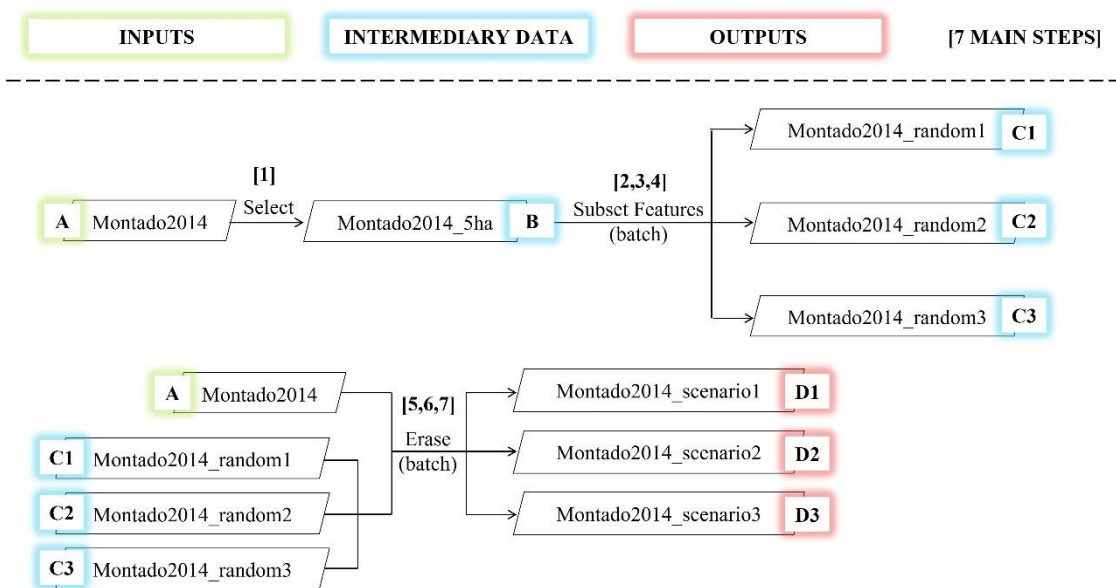
Figure 31 – Scenario building procedure with randomly eliminated small patches (3 runs).

4.3.3. Results and Discussion

4.3.3.1. *Montado* maps

The *montado* area has been decreasing and the NP increasing, which suggests an overall dynamic of fragmentation due to area loss. This is verifiable between 1984 and 1999, 1999 and 2014, and consequently along the entire study period 1984-2014 (Table 9 and Figure 32).

Table 9 – *Montado* number of patches, area and types of dynamic.

	1984	*1999*	*2014*
NP	5.411	5.603	6.131
Area	426.062	401.118	327.178
General ToD	Fragmentation by Loss	Fragmentation by Loss	
	Fragmentation by Loss		
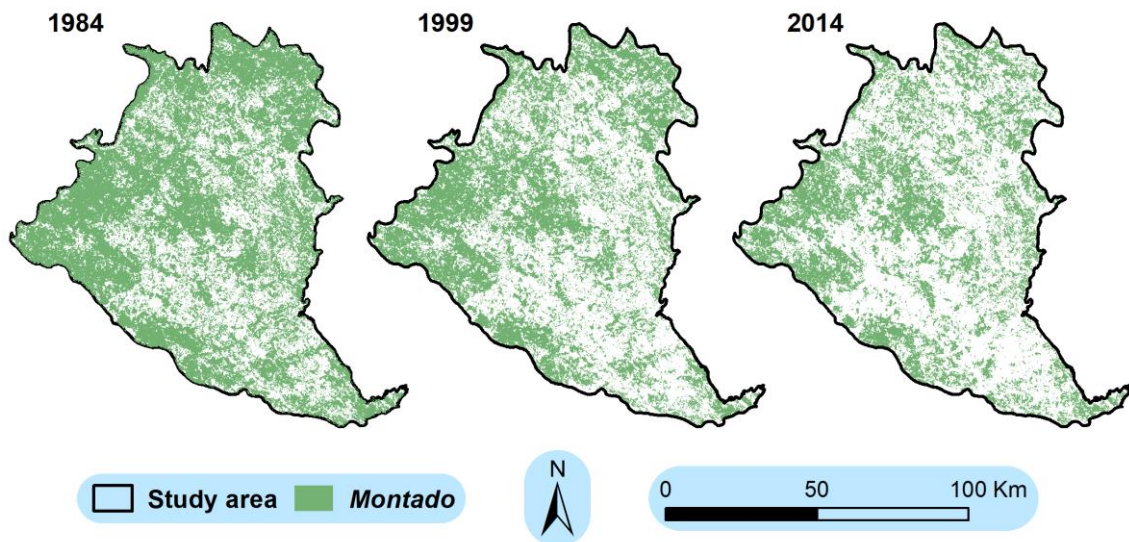
Figure 32 – *Montado* maps for 1984, 1999 and 2014.

In both 1984 and 1999 the existing NP display a left-skewed distribution while the correspondent area displays a right-skewed distribution. These characteristics are also noticeable when the data is aggregated in size classes of 5 ha range (Figure 33). In both dates, the five larger patches are much bigger than the rest and the two larger patches comprise about 55% of the total *montado* area in 1984 and 54% in 1999. Using only the size classes up to 130 ha (black ellipsis in Figure 33), because no larger patches were lost, we have a comparative baseline to assess the losses. Both charts in Figure 34 show that most of the patches are smaller than 5 ha and that this size class also holds the higher area amount.

a)

Montado 1984 - Number of patches and accumulated area per size class (5ha range)

Mont1984	
OBJECTID *	**Hectares**
5315	138086,24976
5206	98126,489008
5396	32090,215708
5271	17048,419244
5294	12983,817235

b)

Montado 1999 - Number of patches and accumulated area per area class (5ha range)

Mont1999	
OBJECTID *	**Hectares**
5511	160568,6
5367	56367,64
5570	29276,36
5341	18946,58
5485	8127,255

Figure 33 – *Montado* number of patches and area per class area; a) in 1984; b) in 1999.
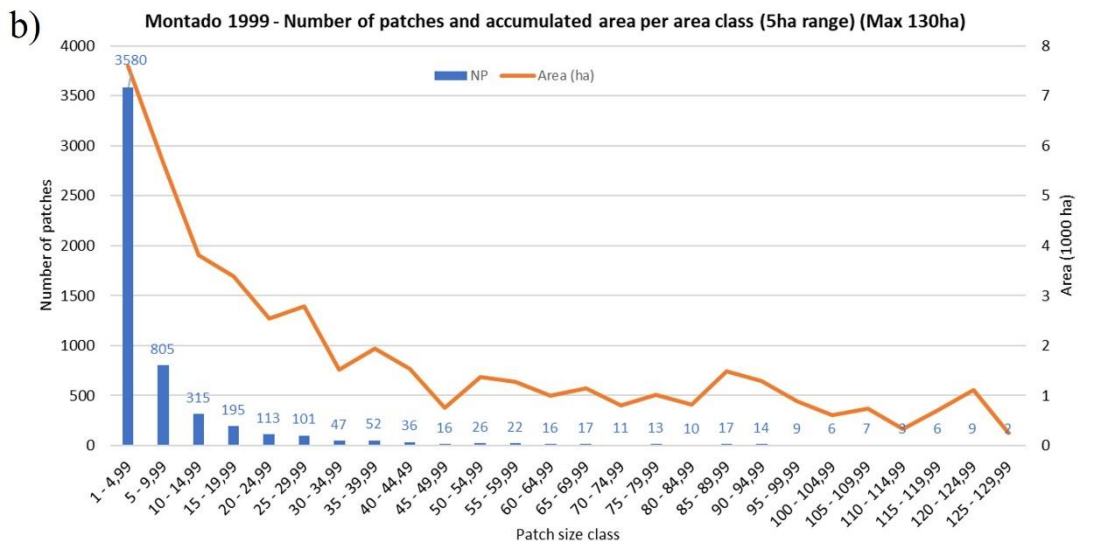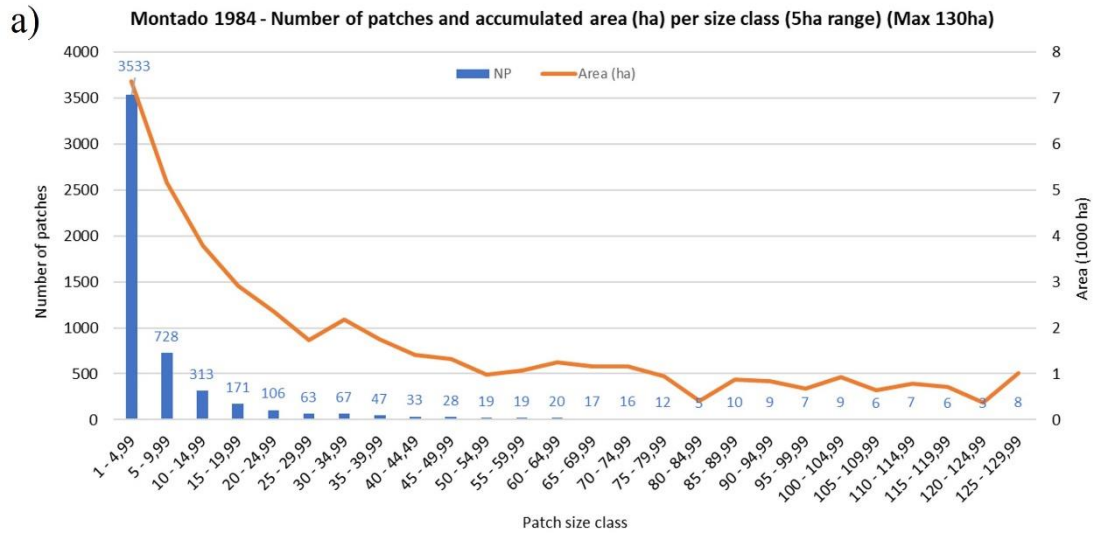
Figure 34 – *Montado* number of patches and area per class area to the maximum of 130 ha; a) in 1984; b) in 1999.

4.3.3.2. *Montado* dynamics

In 1984, 97,25% of the patches were smaller than 130 ha, and these patches held 10,28% of the total *montado* area. The equivalent values for 1999 are 97,23% and 11,56%. Regarding the losses, there was a clear trend showing that smaller patches were lost in higher numbers (Figure 35), while generally, the bigger patches disappeared less often.
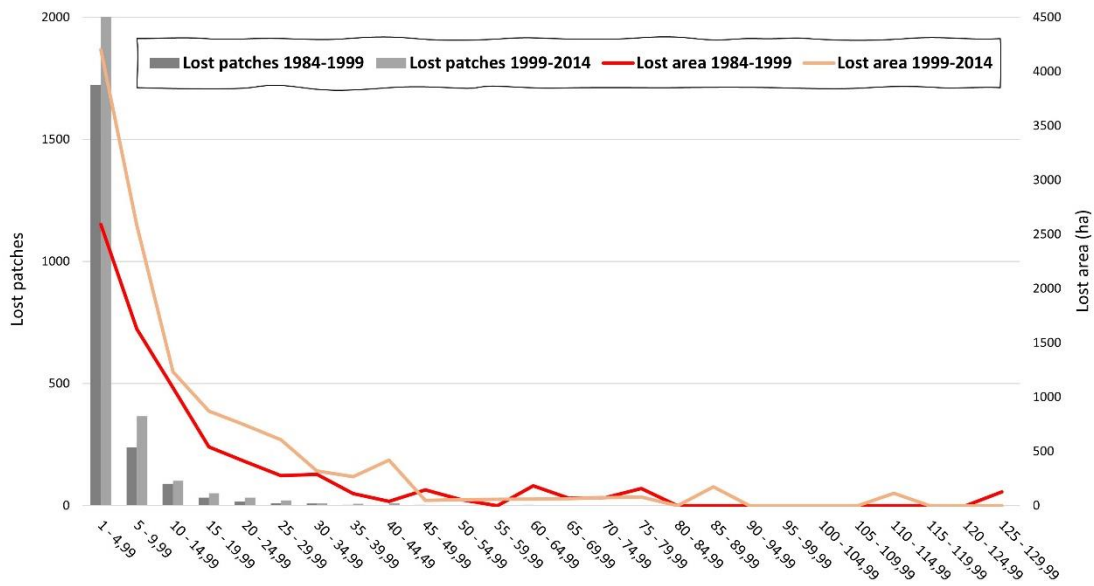
Figure 35 – *Montado* lost patches and lost area in the periods 1984-1999 and 1999-2014.

The existing NP and the lost NP of the different size classes are strongly correlated (Figure 36), which confirms that there are many small patches and many of them are lost and there are few larger patches and few of them are lost. However, the answer we were looking for was if patches within a certain patch size were lost as expected. For example, there are more smaller patches, more smaller patches are lost, but are they disappearing more (proportion-wise) than expected? The Chi-square tests conducted with the most relevant five patch size classes (up to 24,99 ha) show there are differences between size classes, regarding the proportions of existing and lost patches (Box 3 and Figure 37).
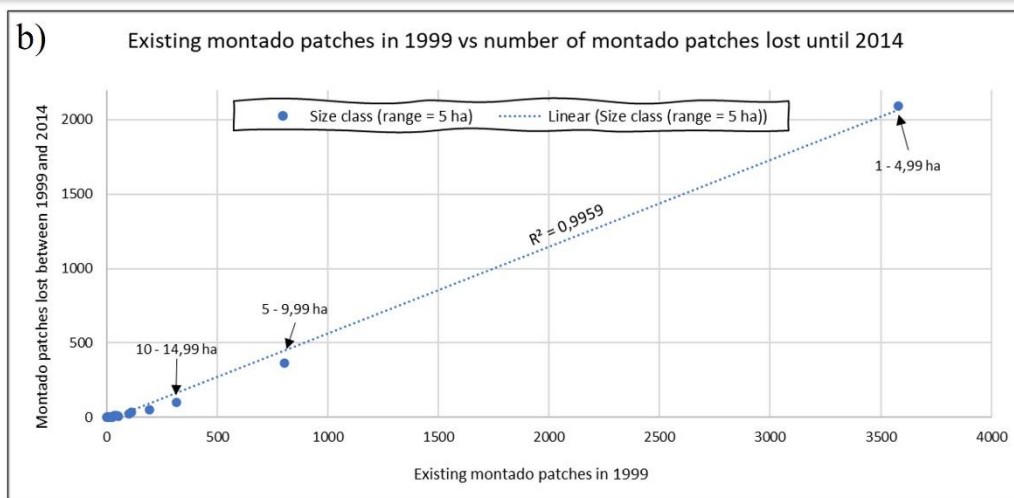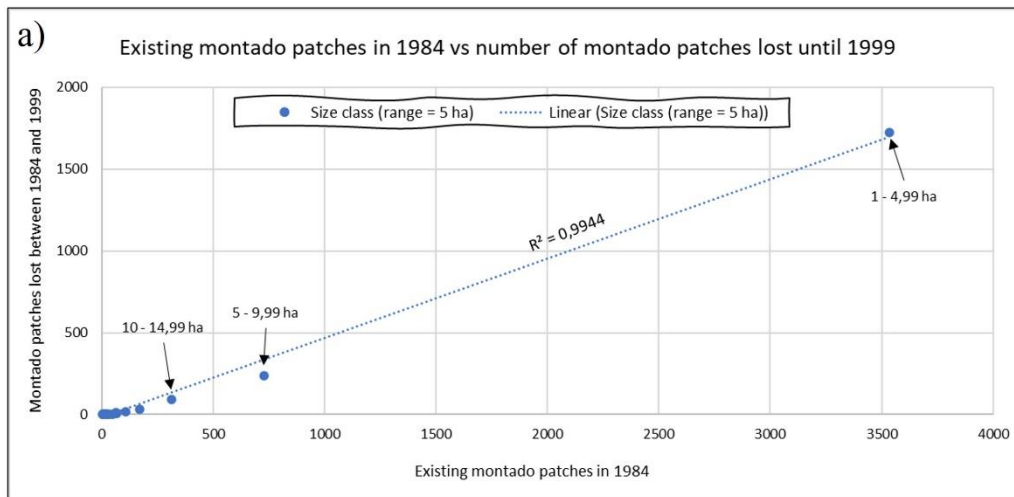
Figure 36 – *Montado* patches vs. lost *montado* patches; a) period 1984-1999; b) period 1999-

2014.

Box 3 – Chi-square test results.

Eight Chi-square tests were performed. Four considering only the five most relevant patch size classes
and four using all the patch size classes. Using all the patch size classes, one criterion to use confidently
the Chi-square test (No expected values should be zero) is not fulfilled. Therefore, Monte Carlo
simulations with 1000 iterations were integrated in the test to simulate the p-value.

For all tests, χ^2 > Critical Value, and also p-value < 0,05, which means we reject H_0, therefore considering
the observed and the expected values differ.

Test 1 – Area1984_5classes
> chisq.test(Area1984_5classes, correct = TRUE)
Pearson's Chi-squared test
data: Area1984_5classes
X-squared = 268;45; df = 4; p-value < 2,2e-16

Test 2 – NP1984_5classes
> chisq.test(NP1984_5classes, correct = TRUE)
Pearson's Chi-squared test
data: NP1984_5classes

X-squared = 76,741; df = 4; p-value = 8,534e-16

Test 3 – Area1999_5classes
> chisq.test(Area1999_5classes, correct = TRUE)
Pearson's Chi-squared test
data: Area1999_5classes
X-squared = 529,34; df = 4; p-value < 2,2e-16

Test 4 – NP1999_5classes
> chisq.test(NP1999_5classes, correct = TRUE)
Pearson's Chi-squared test
data: NP1999_5classes
X-squared = 69,435; df = 4; p-value = 2,988e-14

Test 5 – Area1984_all values
> chisq.test(Area1984, simulate.p.value = TRUE, B = 1000)
Pearson's Chi-squared test with simulated p-value (based on 1000 replicates)
data: Area1984
X-squared = 3463,3; df = NA; p-value = 0,000999

Test 6 – NP1984_all values
> chisq.test(NP1984, simulate.p.value = TRUE, B = 1000)
Pearson's Chi-squared test with simulated p-value (based on 1000 replicates)
data: NP1984
X-squared = 186,59; df = NA; p-value = 0,000999

Test 7 – Area1999_all values
> chisq.test(Area1999, simulate.p.value = TRUE, B = 1000)
Pearson's Chi-squared test with simulated p-value (based on 1000 replicates)
data: Area1999
X-squared = 5253,5; df = NA; p-value = 0,000999

Test 8 – NP1999_all values
> chisq.test(NP1999, simulate.p.value = TRUE, B = 1000)
Pearson's Chi-squared test with simulated p-value (based on 1000 replicates)
data: NP1999
X-squared = 191,9, df = NA; p-value = 0,000999
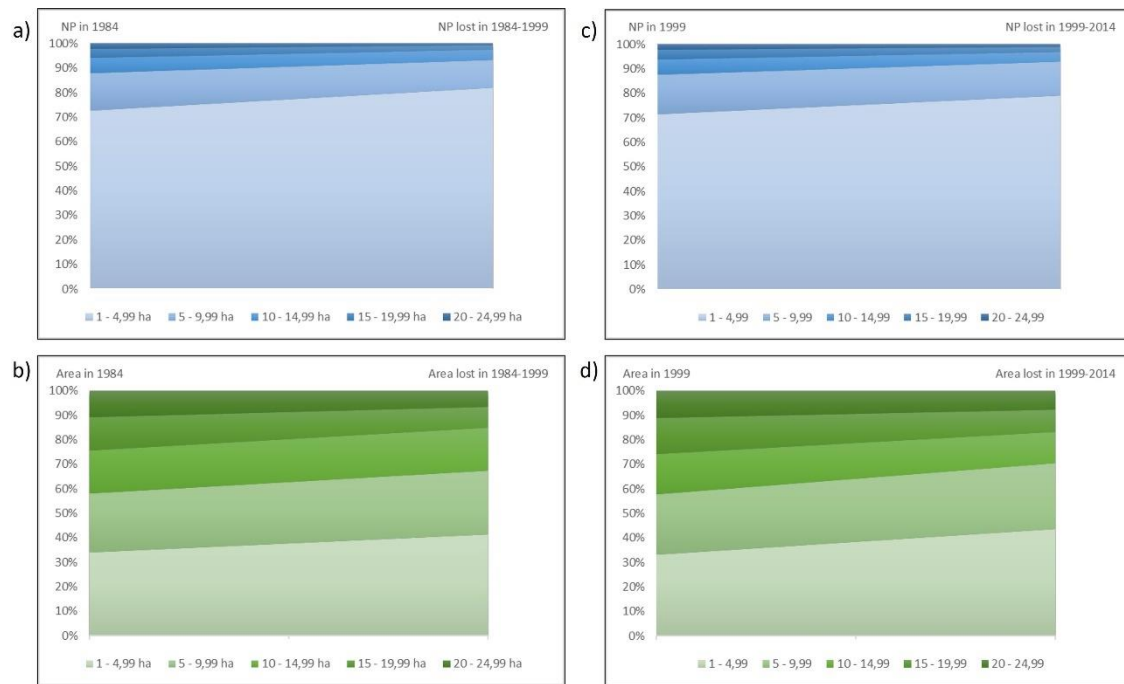
Figure 37 – Percentages of existing NP and Area and Percentages of lost NP and lost Area, grouped by patch size class. a) NP in 1984 and NP lost in 1984-1999; b) Area in 1984 and Area lost in 1984-1999; c) NP in 1999 and NP lost in 1999-2014; d) Area in 1999 and Area lost in 1999-2014.

In Figure 37, charts a) and b) refer to the period 1984-1999 while charts c) and d) refer to the period 1999-2014. The changing heights in the patch size classes show different proportions of existing and the lost patches. In the charts a) and c), concerning NP, it is noticeable that the existing NP of the smaller sizes class were around 70% of the total NP but the percentage of the same class regarding lost patches in the following periods is around 80%. In sum, the smaller patches represented ~70% of the existing patches, but ~80% of the lost patches, leading to the conclusion that patches smaller than 5 ha disappear more than expected. In the next class (5-9,99 ha) the trend is inverted and continues in the next classes, suggesting that patches larger than 5 ha disappear slightly less than expected. The charts b) and d), show area instead of NP and are visually easier to interpret due to the classes' heights. When compared to the charts with NP, they differ mostly in the percentages but do no change much in terms of trends. The patches smaller than 5 ha represent around 30% of the total area but around 40% of the lost area, meaning once again that these small patches disappear more than the initial proportion would lead us to expect. The next size class still shows the same trend

but with a smaller difference, so patches between 5 and 10 ha also disappear more than expected but not as much. The trend inverts for the patches larger than 10 ha, whose extinction occur in lower percentages than their existence. By applying the LDTtool, we obtained additional data regarding local dynamics in 1 km x 1 km squares (Figure 38).
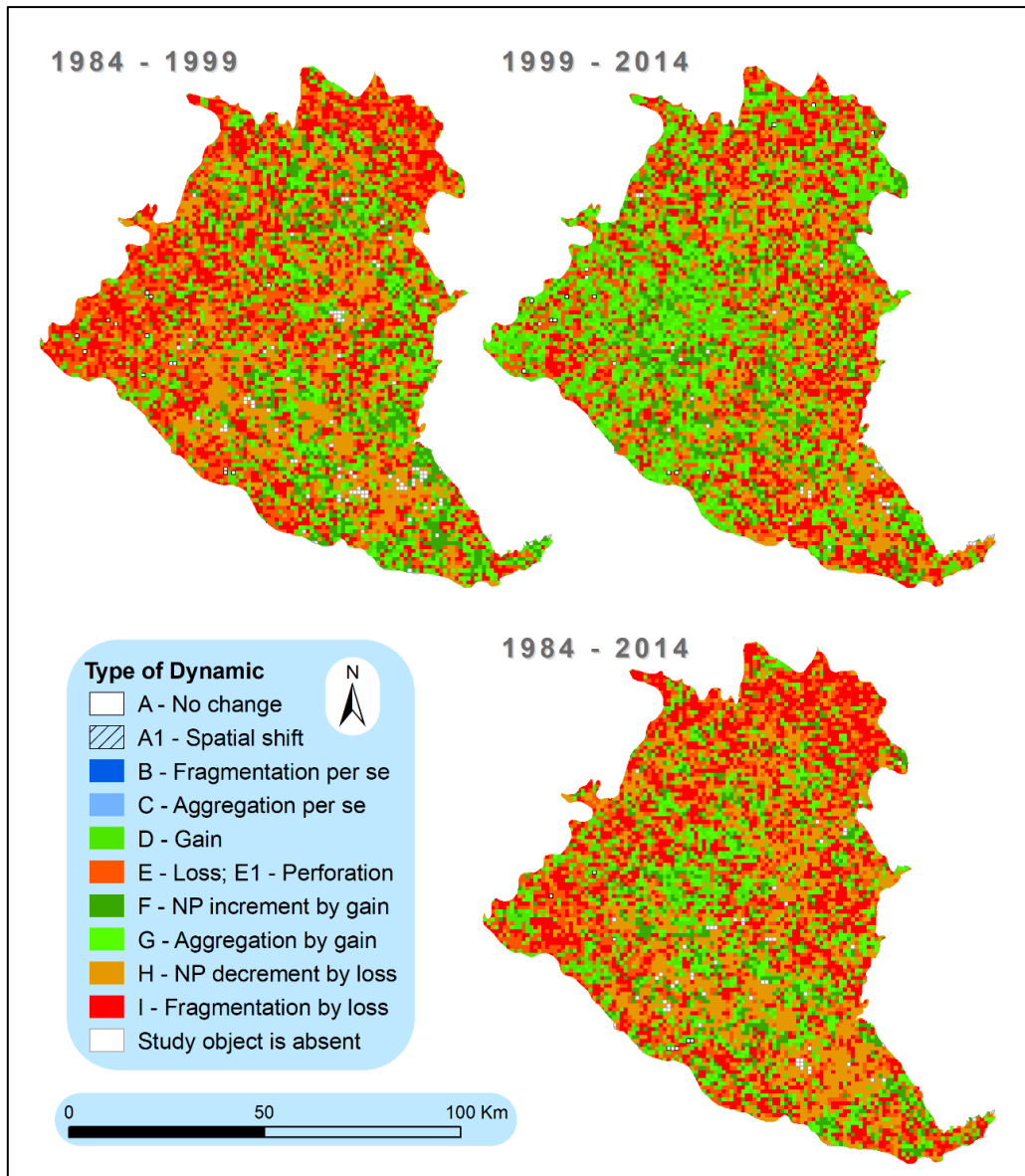
Figure 38 – Types of Dynamic *montado* maps with 1 km x 1 km squares.

Around half (49%) of the patches smaller than 5 ha that existed in 1984 were lost until 1999, which represents 35% of the area comprised in that size class (Table 10). Although the analysis is focused on the lost patches, it is useful to look at the whole picture and realize that other dynamics were at play at the same time. Such a significant

loss of patches could lead us to expect that at the end of the period there would be less of them, but there were actually more (NP = 3.580), and that size class encompasses even more area than before (7.595 ha). Almost half of the small patches disappeared, but new ones also appeared, some by area gain but most due to the 'Fragmentation by loss' suffered by other larger patches. These are two seemingly opposite realities that are in fact related and share the same driver: *montado* loss. In many cases it is the same phenomena, just happening in different size classes, in some provoking NP decrement (small patches disappear altogether) and in other causing NP increment (larger patches break apart). In the period 1999-2014 the percentage of NP and area lost, relative to the size class's total is even higher, reaching 59% and 55%, respectively (Table 11).

Table 10 – Percentage of lost patches and corresponding area in the period 1984-1999 relative to the existing in 1984.

	1984		*Lost in 1984-1999*			
Size class (ha)	*NP*	*Area (ha)*	*NP*	*Area (ha)*	*% NP*	*% Area*
1 - 4,99	3533	7364	1723	2592	49	35
5 - 9,99	728	5163	239	1627	33	32
10 - 14,99	313	3794	90	1087	29	29
15 - 19,99	171	2910	32	541	19	19
20 - 24,99	106	2360	18	407	17	17

Table 11 – Percentage of lost patches and corresponding area in the period 1999-2014 relative to the existing in 1999.

	1999		*Lost in 1999-2014*			
Size class (ha)	*NP*	*Area (ha)*	*NP*	*Area (ha)*	*% NP*	*% Area*
1 - 4,99	3580	7595	2096	4201	59	55
5 - 9,99	805	5676	366	2583	45	46
10 - 14,99	315	3811	102	1232	32	32
15 - 19,99	195	3389	50	871	26	26
20 - 24,99	113	2539	33	742	29	29

4.3.3.3. Patch loss simulation

According to the simulations, losing half of the small patches from 2014 onwards, implies losing around 4.000 ha more (Table 12).

Table 12 – *Montado* loss simulation results. *half of the total NP in 2014, following previous periods patterns.

Size class (ha)	2014 NP	2014 Area (ha)	Loss in Scenario 1 NP	Loss in Scenario 1 Area (ha)	Loss in Scenario 1 % NP	Loss in Scenario 1 % Area
			1934*	4116	50*	50,39

Size class (ha)	NP	Area (ha)	Loss in Scenario 2 NP	Loss in Scenario 2 Area (ha)	Loss in Scenario 2 % NP	Loss in Scenario 2 % Area
1 - 4,99	3868	8169	1934*	4075	50*	49,88

			Loss in Scenario 3 NP	Loss in Scenario 3 Area (ha)	Loss in Scenario 3 % NP	Loss in Scenario 3 % Area
			1934*	4076	50*	49,90

Limiting the analysis to the lost patches smaller than 5 ha, the losses occurred in 1.665 squares in the first period, and 2.013 in the second. Six different ToD were assigned to the squares where the loss of patches smaller than 5 ha took place. The losses are accounted for everywhere but have more influence in some squares than others. Ultimately, it depends on the magnitude and on the dynamics in the surroundings (within the same AU). If nothing more significant happens besides one or more patches being lost, that change causes 'ToD H – NP decrement by loss'. In other cases, despite patches being lost, other spatial patterns mask and overcome that dynamic (e.g., more newly gained patches than lost patches). For this reason, lost patches affect every square but are more significant for ToD H. In the first period, 44% of the squares where these patches were lost displayed ToD H. In the second period, the equivalent value reached 55% (Table 13).

The ToD H counts for the three scenarios are 2.112, 2.090 and 2.077, respectively. However, in the previous periods, only half of the squares in which small patch loss occurred were assigned with ToD H (44% in 1984-1999 and 55% in 1999-2014; average

= 50%). Applying the same proportion, we end up with the ToD H counts being 1.056, 1.045, and 1.039. These are the numbers of squares where the patch loss is expected to be the only or the dominant transformation taking place.

Table 13 – Number of squares assigned to each type of dynamic.

Type of Dynamic	1984-1999		1999-2014	
	Squares (No.)	%	Squares (No.)	%
D – Gain	106	6	120	6
E – Loss	184	11	189	9
F - NP increment by gain	119	7	141	7
G - Aggregation by gain	341	20	285	14
H - NP decrement by loss	736	44	1101	55
I - Fragmentation by loss	179	11	177	9
	Total = 1665		Total = 2013	

4.3.4. Conclusions

The anticipation, based on the empiric experience, that *montado* smaller patches have been disappearing more than larger ones was confirmed. Moreover, the loss of smaller patches is not only disproportionally higher when compared with larger ones but has also accelerated during the analysed period. Conservative scenarios, built on the premise that the trend remains, point to a value of about 4.000 ha reduction, achieved by the loss of patches smaller than 5 ha. If the same amount were lost through large patches, the configurational impact would be greatly reduced but via small patches it originates significant geometric changes in the landscape. In this concern, the scenarios anticipate slightly above 1.000 locations (squares) where the patch loss could be the most relevant dynamic, causing 'ToD H – NP decrement by loss'. This approach is exclusively focused on patch loss, modelled based on past statistics and does not consider *montado* gains or other loss related patterns ('ToD E – Loss' or 'shrinkage'). For that reason, to draw holistic conclusions regarding the *montado* dynamics, would be not only out of the scope of this work, but also too much speculative. An important

improvement to this approach would be to enrich the scenario-building procedure with relevant data concerning the lost patches. For instance, patch isolation or the distance to the nearest patch can influence the patch lost process; thus, such metrics could help unveil some spatial patterns, which in turn would allow us to integrate spatial information in the patch selection for remotion, instead of just randomly removing a portion of the patches.

Since *montado* is a human-made and maintained system, highly dependent on management decisions and actively managed, the conclusions extracted should not be interpreted only as a diagnostic or loose pieces of information to contemplate, while the landscape changes. Notwithstanding more suited proposals for concrete contexts, some general recommendations that stem from our analysis are the following:

- New isolated *montado* patches should cover at least 10 ha and never less than 5 ha.

- If an area increment is smaller than 5 ha, it should be adjacent to existing *montado* patches (Promote 'Gain' or 'Aggregation by gain', instead of 'NP increment by gain').

- If, in a conservation program, other criteria (e.g., interconnectivity being more important than intraconnectivity, connector preservation (Saura and Rubio 2010), etc.)) lead to the preservation of many small patches instead of fewer larger ones, be aware beforehand that the operation will be more demanding and will require more thorough monitoring.

The complex *montado* dynamics are not only subject to natural processes but also very deeply dependent on human decisions that should be supported by the best and more comprehensive information available. Whether it is for *montado* preservation, recovery, or promotion, knowing the patch loss trajectory is vital to outline effective approaches and leverage resources. The confirmation that smaller patches do indeed disappear more, in a disproportioned way and recently at an accelerated rate, is relevant information that should integrate policies and plans, and be reflected in concrete measures.

4.4. Uses by other authors

The fact that LDTtool was used by other authors soon after being released suggests the usability of such tool, if not its necessity. LDTtool was used by Stoica et al. (2021) and Jatayu et al. (2022) to study urban growth and associated dynamics, and by Navarro-Cerrillo et al. (2022) to assess afforestation on agricultural lands.

4.4.1. Built-up area expansion in Romania

The work by Stoica et al. (2021) focuses on urban expansion in the surroundings of Bucharest, Romania, and uses LDTtool to identify the types of build-up area dynamics that occurred between 2000 and 2018. The following excerpts of their discussion highlight the usefulness of LDTtool.

Excerpt 1:

"In addition, LDTtool provides valuable findings. Through quantitative measurements, it identifies the specific dynamics of built-up area and the overall fragmented spatial way of development hinted in other studies [57,75]. Still, some interesting differentiations were revealed, as the prevailing change pattern was "F—NP increment by gain" before 2008, but afterwards the "G—Aggregation by gain" type of dynamics became the most represented, followed by the previously dominant category. Thus, we notice an obvious shift in the last decade toward a spatial homogenization in several places, as the fusion of patches fosters this process [106]."

Excerpt 2:

"For getting deeper insights about territorial transformations, LDTtool has proven suitable for detecting the types of local dynamics, as suggested in previous research [106,107]."

The outputs produced based on LDTtool were a map (Figure 39) and a chart (Figure 40).

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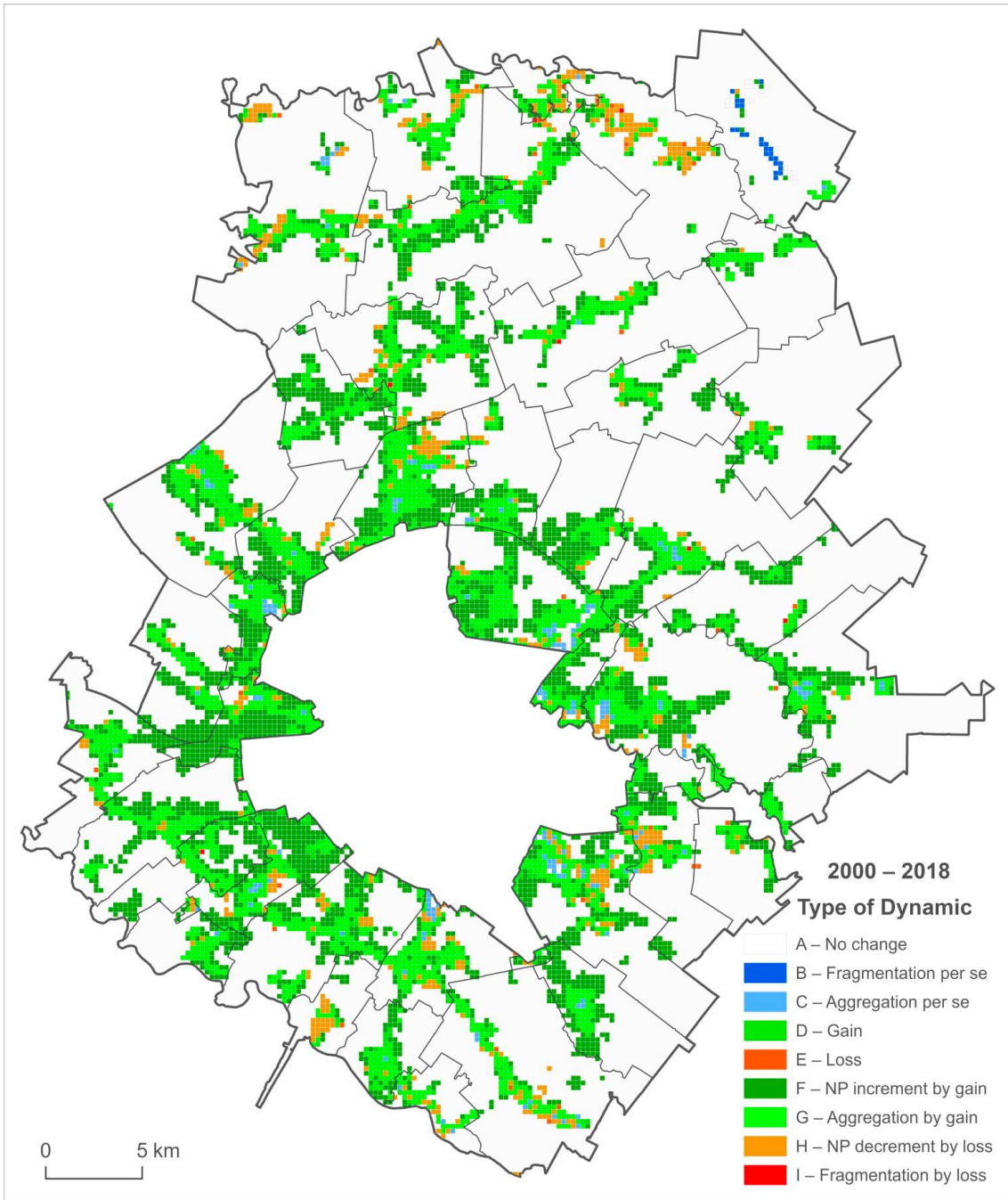
Figure 39 – Built-up area types of dynamics (2000–2018) (Stoica et al. 2021).
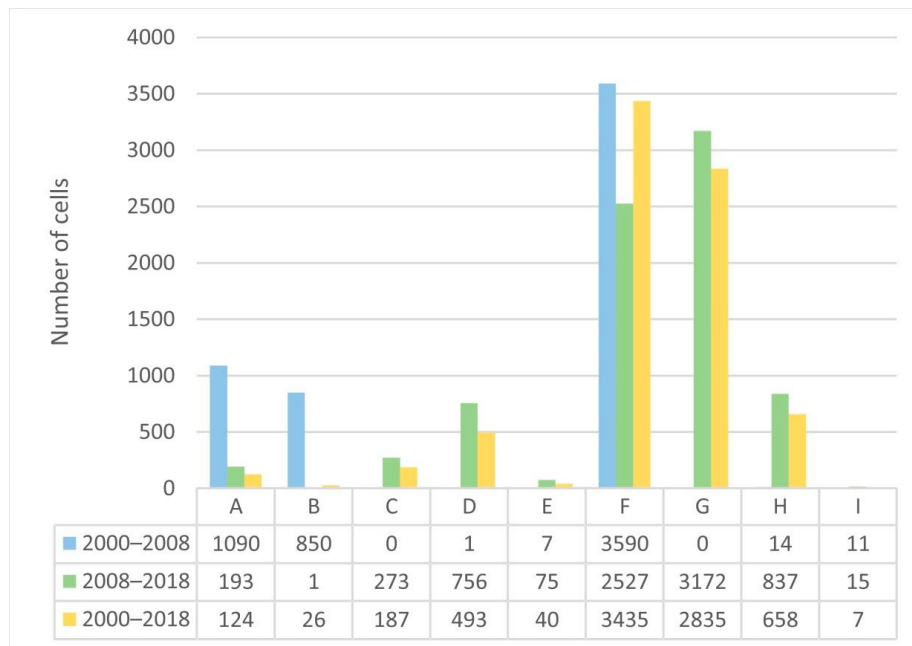
A—No change; B—Fragmentation per se; C—Aggregation per se; D—Gain; E—Loss; F—NP increment by gain; G—Aggregation by gain; H—NP decrement by loss; I—Fragmentation by loss (according to Machado et al 2020)

Figure 40 – Number of cells assigned to each built-up area dynamic type (Stoica et al. 2021).

Reference:

Stoica, I.V., Zamfir, D., Vîrghileanu, M., 2021. Evaluating the territorial impact of built-up area expansion in the surroundings of Bucharest (Romania) through a multilevel approach based on landsat satellite imagery. Remote Sensing. 13. https://doi.org/10.3390/rs13193969

4.4.2. Urban form dynamics in Indonesia

Jatayu et al., (2022) studied the dynamics of urban form (spatial patterns of human activity) in Indonesia. Among other methods, they used the LDT principles, operationalised via LDTtool to assess built-up changes in composition and configuration. The following excerpt is taken from the subsection '2.3.3 Landscape Dynamic Typology (LDT) Tools':

"There are three main interpretations in LDT:
• In a case where there are no changes in area and NP between two periods, the landscape did not experience any dynamics.

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• In a case where the area remains the same between two periods while NP changes, the landscape faces a fragmentation or aggregation event (i.e., pure geometric variation).

• In a case where both the area and NP change between two periods, the landscape is experiencing fragmentation or aggregation due to gain or loss in the specified class (i.e., geometric variation caused by area variation)."

Based on these LDT core principles, they applied LDTtool to the build-up land use category and obtained the output shown in Figure 41, which includes elements of the existing trend and three scenarios. For each one, there is a map dedicated to amount variation, showing urban area gains and losses, and another map showing the geometric variations in the forms of fragmentation or aggregation.
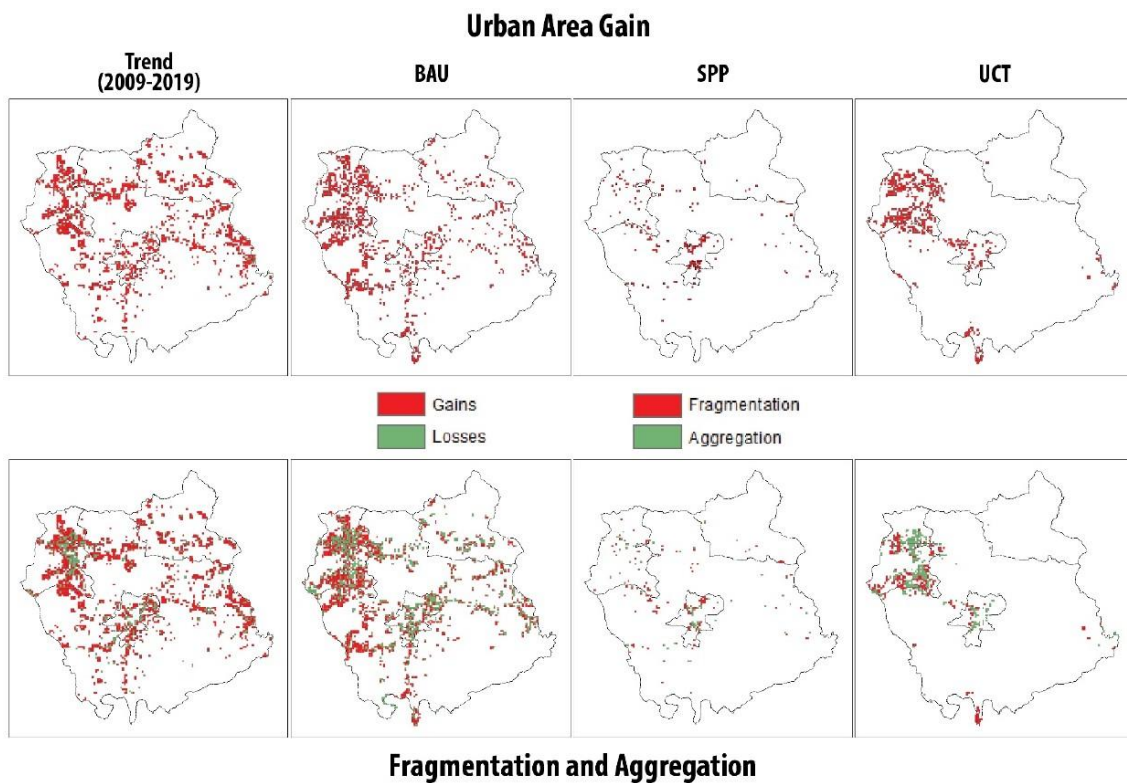
Figure 41 – Spatial distributions of different landscape dynamics identified by trend: BAU, SPP, and UCT models. (Jatayu et al. 2022)

Reference:

Jatayu, A., Saizen, I., Rustiadi, E., Pribadi, D.O., Juanda, B., 2022. Urban Form Dynamics and Modelling towards Sustainable Hinterland Development in North Cianjur, Jakarta–Bandung Mega-Urban Region. Sustainability. 14. https://doi.org/10.3390/su14020907

4.4.3. Afforestation in Southern Spain

The work by Navarro-Cerrillo et al. (2022) focused on landscape fragmentation and connectivity associated with the afforestation on agricultural land between 1990 and 2018. LDTtool was one of the tools used alongside Patch Analyst (Rempel et al. 2012) and Graphab (Foltête et al. 2012). The outputs produced by LDTtool were a group of maps showing the occurred ToD (Figure 42) and a table concerning the area assigned to each ToD, for several combinations of *dehesa*, pine plantation and afforestation.
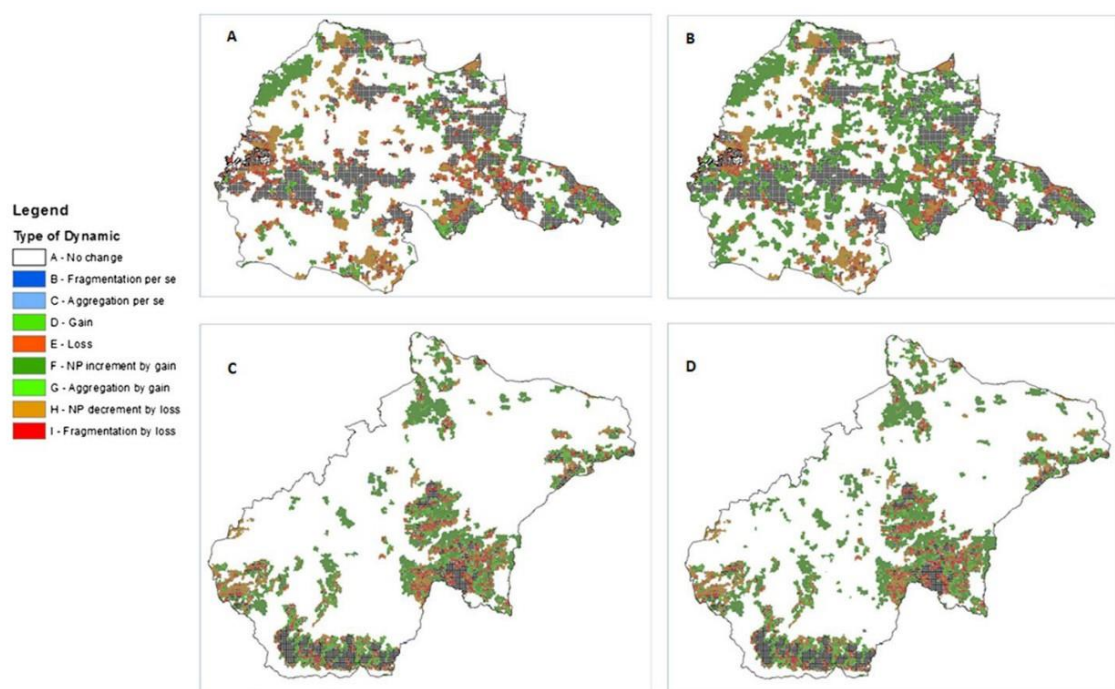
Figure 42 - Dynamic between 1990 and 2018 in *dehesas* and afforestation in agricultural lands in Andevalo (A, B) and *Pinus* and afforestation in agricultural lands in Guadix (C, D), South Spain. (Navarro-Cerrillo et al. 2022)

An excerpt from the article's Results section (Fragmentation dynamic subsection):

"… the ToD "Number of patches (NP) increments by gain" was the most prevalent change pattern in 2018, and it was also the most represented ToD after afforestation (dehesas1990 = 472,43 ha; dehesas + afforestation2018 = 30.399,68 ha). This represents the afforestation were established adjacent to existing forest uses (dehesas)."

Parallel remarks are made concerning other ToD and land uses. Further analysis allowed the authors to draw significant conclusions:

• Most changes related to afforestation were related to the NP increment by gain while fragmentation *per se* was less relevant, i.e., there was not much fragmentation of the already existing patches but there was a significant increase on the number of small patches of new forest.

• New patches have an impact primarily at the landscape granularity level, directly adding variability to the landscape and increasing its conservation value.

Reference:

Navarro-Cerrillo RM, Rivas CA, Quinto L, et al (2022) Afforestation on agricultural land in southern Spain : an important driver to improve forest landscape connectivity. New Forests. https://doi.org/10.1007/s11056-022-09956-4

The conceptual framework and the LDT method were built roughly simultaneously. Once the conceptual framework was clear, the nomenclature emerged naturally from it. This is a broad nomenclature, not meant to replace more specific ones, but instead able to harmonize and provide integrity. With this foundation established, further advances occurred downstream at the tool development level, originating an ArcGIS toolbox and a set of QGIS scripts. Finally, the LDT's analytical capabilities were put to the test in real studies. This thesis' outputs are enumerated next and graphically illustrated in Figure 43.

Outputs:

- A conceptual framework
- A nomenclature
- A method
- Two tools
- Three use cases
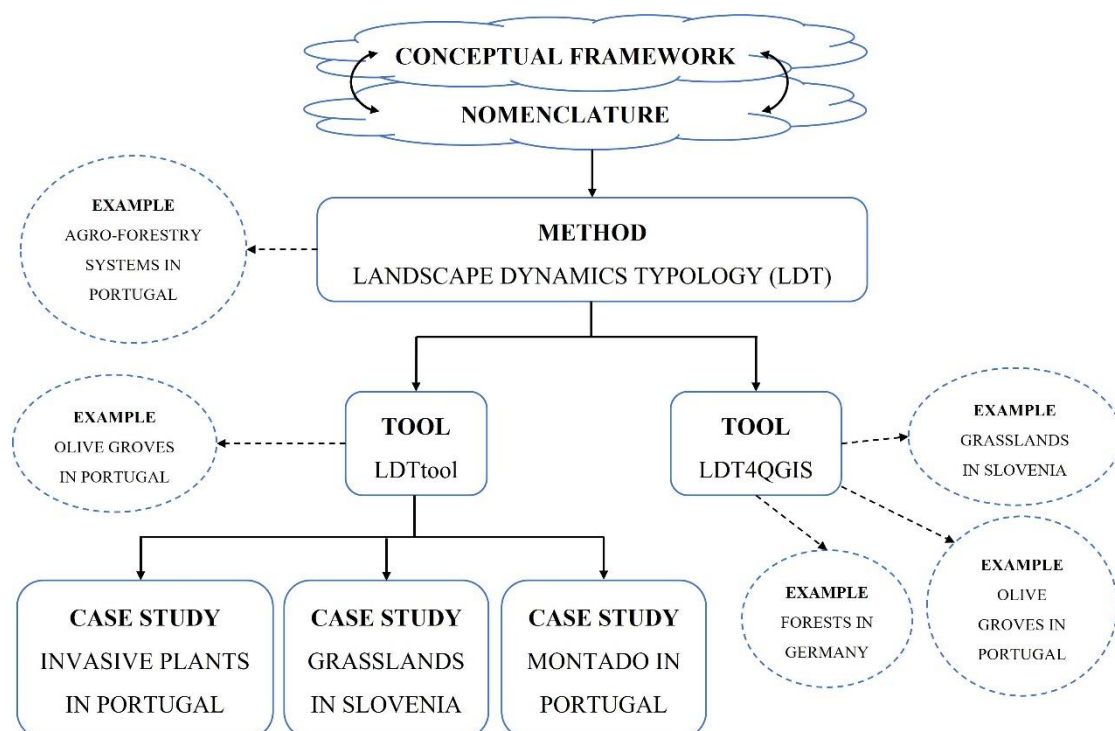
Figure 43 – Graphical overview of the thesis content and outputs.

Neither LDT nor any of the tools are final products. One can think that LDT is solid as it is and requires no further adjustments but any hypothetical improvement in one of the tools (e.g., addition of a ToD) may allow or imply changes in the LDT to assure the overall consistency. LDTtool and LDT4QGIS embody the entire process since they incorporate the conceptual framework, the nomenclature and the method. Their main strengths are the overall user-friendliness, straightforward configuration, possibility to adjust parameters and source code availability that allow users to make their own modifications. The outputs are also easy to interpret as they come in the form of indices variations, and spatially-explicit ToDs/spatial patterns. Both tools' scope of application are limited by certain characteristics such as the use of binary landscapes and artificial AU, which bring together the MAUP. For instance, studies focused on edge contrast metrics or connectivity studies involving resistance surfaces are outside these tools' scopes of application. Correct decisions concerning the data and the phenomena spatial, temporal, and thematic resolutions are needed to assure correct results and conclusions. Caution is also required when interpreting results related to fragmentation, because although the ToD are calculated for each AU, fragmentation is a landscape-scale process (McGarigal and Cushman 2005). Given the variety of topics in which these tools can add analytic and interpretative value (e.g., biodiversity conservation, ecosystem services, landscape planning, etc.), it is likely that original approaches may emerge leading to the development of new features that we cannot yet envision.

The presentation of the method and tools includes demonstrations of how they operate. These illustrative examples also show the type of outputs one can expect. Actual case studies (subchapters 4.1., 4.2. and 4.3.) involved deeper analysis not restricted to LDT but instead adopting a comprehensive approach combining the methods that better fitted the specific needs of each study. In subchapter 4.1. LDTtool was used along with Conefor Sensinode 2.2 (Saura and Torné 2009) to rank *Acacia dealbata* patches based on relevant criteria in order to halt the ongoing invasion. The primordial analysis focused on landscape changes, with connectivity aspects being added to enhance the selection of *Acacia dealbata* patches to eliminate. In this study, LDTtool was applied to build detailed scenarios at the patch level. The analysis performed in subchapter 4.2. regarding the pastures and natural grasslands in Slovenia was distinct because there was no emphasis on specific patches. In this study,

FRAGSTATS (McGarigal and Marks 1995) was used to provide an overall view of the grassland patches nationwide, while LDTtool was used to detect recent changes and to help identify spatial patterns. These are ultimately useful for scenarizing and supporting decision-making on which grasslands should be preserved and actively managed. In subchapter 4.3., the loss of *montado* patches is addressed via LDTtool. Despite adding value through its core capacity of detecting and locating ToDs, the toolbox was specifically used to track lost patches. These were then grouped by size classes, establishing a helpful baseline for predicting which patches are likely to disappear within a short period. In summary, LDT and its associated tools have shown to be able to enrich analytic procedures involving landscape dynamics. In a discipline where topics are often related and intertwined, LDT is easily integrated into broad methodological schemes either feeding from other tools' products (using other software's outputs as inputs) or providing inputs for subsequent analysis (its outputs being inputs for other software).

Zooming out from technicalities and expertise jargon, it is important to keep in mind the end goal is to contribute with methods and tools to improve people's lives through correct landscape planning and natural resources management. Before the advent of GIS, a basic technique of spatial analysis, map overlays, was successfully employed. Map overlays allow visual examination of spatially-explicit characteristics or phenomena, and possible cause-effect relationships between them. The most famous spatial analysis example is the study conducted by the epidemiologist Dr. John Snow on how the cholera outbreak of 1854 in London was related to a specific contaminated water pump (Figure 44).

Figure 44 – Dr. John Snow's map to describe the Broad Street pump cholera outbreak of 1854, in London. (https://www.ph.ucla.edu/epi/snow/mapsbroadstreet.html)

While some issues are more specific and sectorial, others are multidisciplinary and require more complex approaches. Land policies are essential to safeguard the proper use of resources and provide quality of life to populations but can be complicated instruments as they to need to consider many variables and tackle the fact that landscapes are dynamic. Understanding LULC transformations over time is critical for public policies. The opposite is also true since land policies influence land management which directly impacts LULC changes. The strong feedback between land management and landscape dynamics goes beyond environmental traits and is also relevant in terms of socio-economic aspects (Long and Qu 2018). Even intrinsically non-spatial policies and seemingly spatially-agnostic market forces may end up influencing spatial solutions, unintentionally affecting the mosaic of land uses. LULC changes are a spatiotemporal phenomenon and thus can benefit from multiscale analysis. Effective landscape planning requires spatial context of the adjacent landscapes in a region. For example,

irrigated agriculture can be increasing in one municipality but decreasing in the contiguous and stable at the national scale. Even if the study is focused on that one municipality with growing agriculture, it is useful to be aware that the trend is exclusive to it, while in the surrounding municipalities that particular type of agriculture is disappearing. In the same way, landscapes act as sources and sinks, which means, for instance, that flows of fauna from and to the only forested landscape present, affect all the neighboring landscapes.

There is no optimum arrangement of land uses in a landscape as it will vary for several reasons and depend on specific goals, but some general principles seem to be consensual. According to (Forman 1995b), "*the aggregate-with-outliers principle states that one should aggregate land uses, yet maintain corridors and small patches of nature throughout developed areas, as well as outliers of human activity spatially arranged along major boundaries*". Garibaldi et al. (2020) advocate to increase native habitats to at least 20% of working landscape area where the value is currently lower, claiming that such target has benefits for food security and overall nature's contributions to people. Arroyo-Rodríguez et al. (2020) propose landscape scenarios that represent an optimal compromise between delivery of goods and services to humans and preserving forest-dwelling species. To fulfil that goal, landscapes should contain ≥ 40% forest cover (probably higher percentages in the tropics), 10% being comprised in a large patch, and the remaining 30% in many evenly dispersed smaller patches, embedded in a high-quality matrix.

Despite the growing awareness regarding global challenges, local and immediate concerns overwhelmingly determine action. Strategic and coherent regional planning provide context and can help bridge the gap, since the landscape and the region are the central linkage between the global and the local (Forman 1995b). The inherent complexity of LULC changes and the demanding task of landscape design, benefit from clear conceptual frameworks and can be assisted by tools such as the ones developed in this work.

5.1. Future work

A study on landscape dynamics is usually the first step in a more comprehensive effort to gather knowledge about the territory. Besides identifying and characterizing the changes, it is also common to diagnose associated impacts (Alves et al. 2022). In this regard, it would be interesting to quantify the effects of composition and configuration in changing landscapes. LDT identifies situations where changes are purely compositional, purely configurational, or most commonly, a mix of the two, assuming that the compositional variation comes first, hence causing the configurational one. It assesses landscape changes, contributes to distinguish the effects of composition and configuration, but does not offer solutions on how to quantify it. To measure how much of the overall change is due to amount or geometric variations is a much harder task that different authors would probably address in different ways. Since LDT is based on the metrics area and NP, a possible method would be to compare their variations. A large amount variation and a small NP variation would mean that the overall change would be due mainly to composition. Conversely, if a small area variation produced a large NP variation, the emphasis would be put on the configurational part and not so much on the composition. An example of a composition/configuration dominance index is the ratio between the area variation that is associated to NP variation and the total area variation in the AU. The index ranges between 0 and 1: <0,5 - Composition dominance; 0,50 - No dominance; >0,5 - Configuration dominance.

$$COMP/CONF\ Dominance\ Index = \frac{|\Delta Area\ associated\ to\ \Delta NP|}{|\Delta Area|}$$

Example 1: The changes that occurred in a given landscape caused an 8 ha loss, and the 10 patches that were lost covered 6 ha. The major impact is geometrical.

$$COMP/CONF\ Dominance\ Index = \frac{6}{8} = 0,75$$

Example 2: There are two new patches totalling 3 ha but the total area gain is 10 ha. The major impact is compositional.

$$COMP/CONF\ Dominance\ Index = \frac{3}{10} = 0{,}33$$

More complex ToD probably require more sophisticated conceptualization, but this seems to be a good starting point for a line of work that is not necessarily dependent on LDT but could easily enrich it.

Focusing on LDT itself and on the work presented in this thesis, besides the adaptation to R software, other ideas for future improvements are currently being considered. One of them is to allow the selection of the LULC category even if the provided FC/shapefile does not represent a binary landscape. This procedure would eliminate preliminary tasks such as selecting the correct LULC according to its attributes and exporting the polygons into a new FC/shapefile. Furthermore, it would pave the way to expand the analysis beyond binary landscapes or to at least automate it so that it would run sequentially for different LULC based on a single FC/shapefile.

Regarding the delicate task of point aggregation to regularly shaped grids, it is being considered the option of using hexagons as AU. It is a habit and almost a standard procedure to use squares without further thinking or justification. Besides squares, only equilateral triangles and hexagons can tesselate and create a regular grid. However, at least for some applications, such as neighbourhood analysis, movement paths and connectivity, hexagons are more appropriate (Birch et al. 2007). Sampling bias is reduced by low PAR, and since circles and polygons with more than six sides cannot tesselate, hexagons are the best choice.

It has been pointed out that LDTtool and LDT4QGIS, contrary to most landscape analysis software, use the vectorial format and not raster; this has even been framed as an advantage since it may diminish the processing time. Using vector instead of raster happened naturally while the tools were being developed, but there was never a decision to avoid raster format; thus, a raster version can be developed in the future.

Other possible upgrades are:
- Integrate more spatial patterns / ToDs (e.g., dissection, filling (the opposite of perforation), etc.)

- Export a text file containing relevant elements regarding the results (e.g., a summary of ToD counts).

- Fine-tune specific steps in the procedure, such as filtering out spurious polygons caused by overlay operations. A preliminary version of this is actually done but is still not integrated because it is not user-friendly and requires significant explanation regarding settings and thresholds. A better, simpler version is being developed and will be implemented once it is ready.

The software is available for download in the developers' GitHub and GitLab pages. The repositories contain informative and instructional README files. Updates will happen whenever an improved version of the software is ready for release.

LDTtool (Rui Machado's GitHub page)   **LDT4QGIS (Luís Paixão's GitLab page)**

In the period 2018-2022, Rui Machado was also involved in the following works:

Machado R, Santos P (2019) Does wild rabbit population size affect connectivity? World Rabbit Science, 27:207-216. DOI:10.4995/wrs.2019.12068

Guiomar N, Godinho S, Rivera M, Pinto-Correia T, **Machado R**, Czekaj M, Tyran E, Puchała J (2021) Assessing food availability: A novel approach for the quantitative estimation of the contribution of small farms in regional food systems in Europe. Global Food Security 30:100555. DOI: 10.1016/j.gfs.2021.100555

Guiomar, N., Faria, S., Godinho, S., **Machado, R**., Boscolo, D., 2021. Dinâmicas e evolução da paisagem. In: Carvalho-Ribeiro, S., Boscolo, D., Ciochetti, G., Firmino, A., Guiomar, N. (Org.) Ecologia da paisagem no contexto Luso-Brasileiro – Volume I, Editora Appris, Curitiba, pp. 113-150. ISBN: 9786525002798

Acácio V, Dias FS, Catry FX, et al (2021) Canopy Cover Loss of Mediterranean Oak Woodlands : Long-term Effects of Management and Climate. Ecosystems. https://doi.org/10.1007/s10021-021-00617-9

Allen H, Simonson W, Parham E, et al (2018) Satellite remote sensing of land cover change in a mixed agro-silvo-pastoral landscape in the Alentejo, Portugal. Int J Remote Sens 39:4663–4683. https://doi.org/10.1080/01431161.2018.1440095

Almeida M, Azeda C, Guiomar N, Pinto-Correia T (2016) The effects of grazing management in montado fragmentation and heterogeneity. Agrofor Syst 90:69–85. https://doi.org/10.1007/s10457-014-9778-2

Alves A, Marcelino F, Gomes E, et al (2022) Spatiotemporal Land-Use Dynamics in Continental Portugal 1995–2018. Sustainability 14. https://doi.org/https://doi.org/10.3390/su142315540

Arroyo-Rodríguez V, Fahrig L, Tabarelli M, et al (2020) Designing optimal human-modified landscapes for forest biodiversity conservation. Ecol Lett 23:1404–1420. https://doi.org/10.1111/ele.13535

Arsic Z (2015) European landscape convention. Zb. Rad. Pravnog Fak. Novi Sad 49:457–467

Assandri G, Bogliani G, Pedrini P, Brambilla M (2019) Toward the next Common Agricultural Policy reform: Determinants of avian communities in hay meadows reveal current policy's inadequacy for biodiversity conservation in grassland ecosystems. J Appl Ecol 56:604–617. https://doi.org/10.1111/1365-2664.13332

Birch CPD, Oom SP, Beecham JA (2007) Rectangular and hexagonal grids used for observation, experiment and simulation in ecology. Ecol Modell 206:347–359. https://doi.org/10.1016/j.ecolmodel.2007.03.041

Blondel J (2006) The 'Design' of Mediterranean Landscapes: A Millennial Story of Humans and Ecological Systems during the Historic Period. Hum Ecol 34:713–729. https://doi.org/10.1007/s10745-006-9030-4

Bogaert J, Ceulemans R, Salvador-Van Eysenrode D (2004) Decision Tree Algorithm for

Detection of Spatial Processes in Landscape Transformation. Environ Manage 33:62–73. https://doi.org/10.1007/s00267-003-0027-0

Bolliger J, Kienast F (2010) Landscape functions in a changing environment. Landsc Online 21:1–5. https://doi.org/10.3097/LO.201021

Broyer J, Pelloli L, Curtet L, Chazal R (2017) On habitat characteristics driving meadow passerine densities in lowland hay-meadow systems in France. Agric Ecosyst Environ 237:24–30. https://doi.org/10.1016/j.agee.2016.12.021

Bugalho M, Plieninger T, Aronson J, et al (2009) Open Woodlands: A Diversity of Uses (and Overuses). In: Aronson J, Pereira JS, Pausas JG (eds) Cork Oak Woodlands on the Edge, Ecology, Adaptive Management, and Restoration, 1st edn. Society for Ecological Restoration International, Island Press, Washington D.C., pp 33–45

Bugalho MN, Caldeira MC, Pereira JS, et al (2011) Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Front Ecol Environ 9:278–286. https://doi.org/10.1890/100084

Butkiewicz T, Dou W, Wartell Z, et al (2008) Multi-Focused Geospatial Analysis Using Probes. IEEE Trans Vis Comput Graph 14:1165–1172

Butkiewicz T, Meentemeyer RK, Shoemaker DA, et al (2010) Alleviating the modifiable areal unit problem within probe-based geospatial analyses. Comput Graph Forum 29:923–932. https://doi.org/10.1111/j.1467-8659.2009.01707.x

Cadenasso ML, Pickett STA (2001) Effect of Edge Structure on the Flux of Species into Forest Interiors. Conserv Biol 15:91–97. https://doi.org/10.1111/j.1523-1739.2001.99309.x

Caetano M, Nunes V, Nunes A (2009) CORINE Land Cover 2006 for Continental Portugal, Relatório técnico. [CORINE Land Cover 2006 for Continental Portugal, technical report]

Callaway RM, Aschehoug ET (2000) Invasive plants versus their new and old neighbors: A mechanism for exotic invasion. Science (80-) 290:521–523. https://doi.org/10.1126/science.290.5491.521

Campbell P (1993) Wattle Control. Plant Protec5on Research Ins5tute. Handbook n° 3. Pretoria, South Africa

Cancela D'Abreu A, Pinto-Correia T, Oliveira R (2004) Contributos para a identificação e caracterização da Paisagem em Portugal Continental. Direcção Geral do

Ordenamento do Território e Desenvolvimento Urbano, Lisboa

Canteiro C, Pinto-Cruz C, Simões MP, Gazarini L (2011) Conservation of Mediterranean oak woodlands: Understorey dynamics under different shrub management. Agrofor Syst 82:161–171. https://doi.org/10.1007/s10457-011-9375-6

Cheney C, Esler KJ, Foxcroft LC, van Wilgen NJ (2019) Scenarios for the management of invasive Acacia species in a protected area: Implications of clearing efficacy. J Environ Manage 238:274–282. https://doi.org/10.1016/j.jenvman.2019.02.112

Collinge SK (1996) Ecological consequences of habitat fragmentation: Implications for landscape architecture and planning. Landsc Urban Plan 36:59–77. https://doi.org/10.1016/S0169-2046(96)00341-6

Corbin JD, D'Antonio CM (2012) Gone but Not Forgotten ? Invasive Plants' Legacies on Community and Ecosystem Properties. Invasive Plant Sci Manag 5:117–124. https://doi.org/10.1614/IPSM-D-11-00005.1

Costa JC, Aguiar C, Capelo JH, et al (1998) Biogeografia de Portugal Continental. Quercetea 0:5–56. https://doi.org/citeulike-article-id:9981809

Cousens R, Mortimer M (1995) Dynamics of Weed Populations. Cambridge University Press, Cambridge

Cumming S, Vernier P (2002) Statistical models of landscape pattern metrics, with applications to regional scale dynamic forest simulations. Landsc Ecol 17:433–444. https://doi.org/10.1023/A:1021261815066

Cushman SA, McGarigal K, Neel MC (2008) Parsimony in landscape metrics : Strength , universality , and consistency. Ecol Indic 8:691–703. https://doi.org/10.1016/j.ecolind.2007.12.002

D'Antonio C, Meyerson LA (2002) Exotic plant species as problems and solutions in ecological restoration: A synthesis. Restor Ecol 10:703–713. https://doi.org/10.1046/j.1526-100X.2002.01051.x

Dakskobler I, Poldini L (2019) Alpine grasslands with dominant luzula alpinopilosa in the julian and carnic alps (nw slovenia, ne italy). Hacquetia 18:25–46. https://doi.org/10.2478/hacq-2018-0011

Dakskobler I, Seliškar A (2016) Two new montane grassland communities from the SE Alps (N Slovenia). Hacquetia 15:31–48. https://doi.org/10.1515/hacq-2016-0008

Devictor V, Julliard R, Clavel J, et al (2008) Functional biotic homogenization of bird

communities in disturbed landscapes. Glob Ecol Biogeogr 17:252–261.
https://doi.org/10.1111/j.1466-8238.2007.00364.x

Diagne C, Leroy B, Gozlan RE, et al (2020) InvaCost, a public database of the economic
costs of biological invasions worldwide. Sci Data 7:1–12.
https://doi.org/10.1038/s41597-020-00586-z

Diagne C, Leroy B, Vaissière AC, et al (2021) High and rising economic costs of
biological invasions worldwide. Nature 592:571–576.
https://doi.org/10.1038/s41586-021-03405-6

Dinis C, Surový P, Ribeiro N, Oliveira MRG (2015) The effect of soil compaction at
different depths on cork oak seedling growth. New For 46:235–246.
https://doi.org/10.1007/s11056-014-9458-0

Direção-Geral do Território (2019) Especificações técnicas da Carta de uso e ocupação
do solo (COS) de Portugal Continental para 2018. Relatório Técnico.

Donovan TM, Strong AM (1997) Linkages between Landscape Theory and Population
Dynamics. A Review of Empirical Evidence. In: Bissonette JA, Storch I (eds)
Landscape Ecology and Resource Management: Linking Theory with Practice.
Island Press, Washington, DC, pp 35–45

Erjavec E, Šumrada T, Juvančič L, et al (2018) Vrednotenje slovenske kmetijske politike
v obdobju 2015 – 2020: Raziskovalna podpora za strateško načrtovanje po letu
2020. [Evaluation of the Slovenian Agricultural Policy in 2015-2020 Period.
Research Support for Strategic Planning post-2020]. Kmetijski

Esgalhado C, Guimarães MH, Lardon S, et al (2021) Mediterranean land system
dynamics and their underlying drivers: Stakeholder perception from multiple case
studies. Landsc Urban Plan 213:.
https://doi.org/10.1016/j.landurbplan.2021.104134

ESRI (2013) ArcGIS Release 10.2. Redlands, CA: Environmental Systems Research
Institute

ESRI (2016) ArcGIS Release 10.4.1 Redlands, CA: Environmental Systems Research
Institute

European Environment Agency (2020) State of nature in the EU. Results from reporting
under the nature directives 2013-2018. (EEA Report No. 10/2020). Copenhagen

European Union (2013) Habitat Directive 92/43 EEC. Counc. Eur. Communities Counc.

Dir. 92/43/EEC 21 May 1992 Conserv. Nat. habitats wild fauna flora, Consol. version 01.07.2013

Ewel JJ (1986) Invasibility: Lessons from South Florida. In: Mooney HA, Drake JA (eds) Ecology of biological invasions of North America and Hawaii. Springer-Verlag, New York, pp 214–230

Fahrig L (2003) Effects of Habitat Fragmentation on Biodiversity. Annu Rev Ecol Evol Syst 34:487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Fahrig L (2017) Ecological Responses to Habitat Fragmentation Per Se. Annu Rev Ecol Evol Syst 48:annurev-ecolsys-110316-022612. https://doi.org/10.1146/annurev-ecolsys-110316-022612

Fahrig L (1997) Relative effects of habitat loss and fragmentation on population extinction. J. Wildl. Manage. 61:603–610

Farina A (1998) Principles and Methods in Landscape Ecology. Chapman & Hall, London, UK

Fazey I, Fischer J, Lindenmayer DB (2005) What do conservation biologists publish? Biol Conserv 124:63–73. https://doi.org/10.1016/j.biocon.2005.01.013

Fernández-Habas J, Sánchez-Zamora P, Ceña-Delgado F, Gallardo-Cobos R (2018) Assessment of ecosystem services provision: The case of mountain olive groves in los pedroches, southern Spain. New Medit 17:43–60. https://doi.org/10.30682/nm1802d

Ferreira V, Figueiredo A, Graça MAS, et al (2021) Invasion of temperate deciduous broadleaf forests by N-fixing tree species – consequences for stream ecosystems. Biol Rev. https://doi.org/10.1111/brv.12682

Fischer J, Lindenmayer DB (2007) Landscape modification and habitat fragmentation: a synthesis. Glob Ecol Biogeogr 16:265–280. https://doi.org/10.1111/j.1466-8238.2006.00287.x

Foley JA, Defries R, Asner GP, et al (2005) R EVIEW Global Consequences of Land Use. 8:570–575

Foltête JC, Clauzel C, Vuidel G (2012) A software tool dedicated to the modelling of landscape networks. Environ Model Softw 38:316–327. https://doi.org/10.1016/j.envsoft.2012.07.002

Forman RTT (1995a) Land mosaics - The ecology of landscapes and regions. Cambridge

University Press, Cambridge

Forman RTT (1995b) Some general principles of landscape and regional ecology. Landsc Ecol 10:133–142

Forman RTT, Godron M (1986) Landscape Ecology. Cambridge University Press, Cambridge

Franklin JF, Forman RTT (1987) Creating landscape patterns by forest cutting: Ecological consequences and principles. Landsc Ecol 1:5–18. https://doi.org/10.1007/BF02275261

Friedman JH (2002) Stochastic gradient boosting. Comput Stat Data Anal 38:367–378. https://doi.org/10.1016/S0167-9473(01)00065-2

Gaertner M, Nottebrock H, Fourie H, et al (2012) Plant invasions, restoration, and economics: Perspectives from South African fynbos. Perspect Plant Ecol Evol Syst 14:341–353. https://doi.org/10.1016/j.ppees.2012.05.001

Garibaldi LA, Oddi FJ, Miguez FE, et al (2020) Working landscapes need at least 20% native habitat. Conserv Lett 1–10. https://doi.org/10.1111/conl.12773

Gaspar P, Mesías FJ, Escribano M, et al (2007) Economic and management characterization of dehesa farms: Implications for their sustainability. Agrofor Syst 71:151–162. https://doi.org/10.1007/s10457-007-9081-6

Gehlke CE, Biehl K (1934) Certain Effects of Grouping Upon the Size of the Correlation Coefficient in Census Tract Material. J Am Stat Assoc 29:169–170

Ghersa CM, Benech-Arnold RL, Satorre EH, Martínez-Ghersa MA (2000) Advances in weed management strategies. F Crop Res 67:95–104

Gibson MR, Richardson DM, Marchante E, et al (2011) Reproductive biology of Australian acacias: Important mediator of invasiveness? Divers Distrib 17:911–933. https://doi.org/10.1111/j.1472-4642.2011.00808.x

Gioria M, Pyšek P (2015) The Legacy of Plant Invasions: Changes in the Soil Seed Bank of Invaded Plant Communities. Bioscience 66:40–53

Godinho C, Rabaça JE (2011) Birds like it Corky: The influence of habitat features and management of "montados" in breeding bird communities. Agrofor Syst 82:183–195. https://doi.org/10.1007/s10457-010-9345-4

Godinho S, Gil A, Guiomar N, et al (2016a) A remote sensing-based approach to estimating montado canopy density using the FCD model: a contribution to

identifying HNV farmlands in southern Portugal. Agrofor Syst 90:23–34. https://doi.org/10.1007/s10457-014-9769-3

Godinho S, Guiomar N, Gil A (2016b) Using a stochastic gradient boosting algorithm to analyse the effectiveness of Landsat 8 data for montado land cover mapping: Application in southern Portugal. Int J Appl Earth Obs Geoinf 49:151–162. https://doi.org/10.1016/j.jag.2016.02.008

Godinho S, Guiomar N, Machado R, et al (2016c) Assessment of environment, land management, and spatial variables on recent changes in montado land cover in southern Portugal. Agrofor Syst 90:177–192. https://doi.org/10.1007/s10457-014-9757-7

Gómez JM (2003) Spatial patterns in long-distance dispersal of Quercus ilex acorns by jays in a heterogeneous landscape. Ecography (Cop) 26:573–584. https://doi.org/10.1034/j.1600-0587.2003.03586.x

Gotelli N, Graves G (1996) Null models in ecology. Smithsonian Institution Press, Washington, DC

Gustafson EJ (2019) How has the state-of-the-art for quantification of landscape pattern advanced in the twenty-first century? Landsc Ecol 34:2065–2072. https://doi.org/10.1007/s10980-018-0709-x

Gustafson EJ (1998) Quantifying Landscape Spatial Pattern : What Is the State of the Art? Ecosystems 1:143–156

Haila Y (2002) A Conceptual Genealogy of Fragmentation Research: From Island Biogeography to Landscape Ecology. Ecol Appl 12:321. https://doi.org/10.2307/3060944

Haines-Young R, Chopping M (1996) Quantifying Landscape Structure: A Review of Landscape Indices and Their Application to Forested Landscapes. Prog Phys Geogr 20:418–445

Hall LS, Krausman PR, Morrison ML (1997) The habitat concept and a plea for standard terminology. Wildl Soc Bull 25:173–182

Hargis CD, Bissonette JA, David JL (1998) The behavior of landscape metrics commonly used in the study of habitat fragmentation. Landsc Ecol 13:167–186

Harms RS, Hiebert RD (2006) Vegetation Response Following Invasive Tamarisk (Tamarix spp .) Removal and Implications for Riparian Restoration. Restor Ecol

14:461–472

Hejcman M, Hejcmanová P, Pavlů V, Beneš J (2013) Origin and history of grasslands in central europe - A review. Grass Forage Sci 68:345–363. https://doi.org/10.1111/gfs.12066

Herrera JM, García D (2010) Effects of forest fragmentation on seed dispersal and seedling establishment in ornithochorous trees. Conserv Biol 24:1089–1098. https://doi.org/10.1111/j.1523-1739.2010.01459.x

Hesselbarth M, Sciaini M, With KA, et al (2019) landscapemetrics: an open-source R tool to calculate landscape metrics. Ecography (Cop) 42:1648–1657

Higgins SI, Richardson DM, Cowling RM (2000) Using a dynamic landscape model for planning the management of alien plant invasions. Ecol Appl 10:1833–1848. https://doi.org/10.1890/1051-0761(2000)010[1833:UADLMF]2.0.CO;2

Holmes PM, Esler KJ, van Wilgen BW, Richardson DM (2020) Ecological restoration of ecosystems degraded by invasive alien plants in South African Fynbos: Is spontaneous succession a viable strategy? Trans R Soc South Africa 111–139. https://doi.org/10.1080/0035919X.2020.1781291

Holmes PM, Richardson DM, Van Wilgen BW, Gelderblom C (2000) Recovery of South African fynbos vegetation following alien woody plant clearing and fire: Implications for restoration. Austral Ecol 25:631–639. https://doi.org/10.1111/j.1442-9993.2000.tb00069.x

Hulme PE (2006) Beyond control : wider implications for the management of biological invasions. J Appl Ecol 43:835–847. https://doi.org/10.1111/j.1365-2664.2006.01227.x

IPBES (2019) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. E. S. Brondizio, J. Settele, S. Díaz, and H. T. Ngo (editors). IPBES secretariat, Bonn

Jaeger JAG (2000) Landscape division, splitting index, and effective mesh size: New measures of landscape fragmentation. Landsc Ecol 15:115–130. https://doi.org/10.1023/A:1008129329289

Jaeger JAG, Soukup T, Schwick C, et al (2011) Landscape fragmentation in Europe

Jatayu A, Saizen I, Rustiadi E, et al (2022) Urban Form Dynamics and Modelling towards

Sustainable Hinterland Development in North Cianjur, Jakarta–Bandung Mega-Urban Region. Sustain 14:. https://doi.org/10.3390/su14020907

Johnstone CP, Lill A, Reina RD (2014) Habitat loss, fragmentation and degradation effects on small mammals: Analysis with conditional inference tree statistical modelling. Biol Conserv 176:80–98. https://doi.org/10.1016/j.biocon.2014.04.025

Jones BA (2017) Invasive Species Impacts on Human Well-being Using the Life Satisfaction Index. Ecol Econ 134:250–257. https://doi.org/10.1016/j.ecolecon.2017.01.002

Jugovic J, Koprivnikar N, Koren T (2018) The role of semi–natural grasslands and livestock in sustaining dung beetle communities (Coleoptera, scarabaeoidea) in sub–mediterranean areas of Slovenia. Anim Biodivers Conserv 41:321–332. https://doi.org/10.32800/abc.2018.41.0321

Kaligarič M, Čuš J, Škornik S, Ivajnšič D (2019) The failure of agri-environment measures to promote and conserve grassland biodiversity in Slovenia. Land use policy 80:127–134. https://doi.org/10.1016/j.landusepol.2018.10.013

Kmecl P, Denac K (2018) The effects of forest succession and grazing intensity on bird diversity and the conservation value of a Northern Adriatic karstic landscape. Biodivers Conserv 27:2003–2020. https://doi.org/10.1007/s10531-018-1521-2

Kmecl P, Gamser M, Šumrada T (2020) Monitoring splošno razširjenih vrst ptic za določitev slovenskega indeksa ptic kmetijske krajine - končno poročilo za leto 2020. Monitoring of common bird species for the determination of Slovenian farmland bird index - final report for the year 2020 – DO

Konlechner TM, Hilton MJ, Lord JM (2015) Plant community response following the removal of the invasive Lupinus arboreus in a coastal dune system. Restor Ecol 23:607–614. https://doi.org/10.1111/rec.12234

Krug RM, Roura-Pascual N, Richardson DM (2010) Clearing of invasive alien plants under different budget scenarios: Using a simulation model to test efficiency. Biol Invasions 12:4099–4112. https://doi.org/10.1007/s10530-010-9827-3

Kumschick S, Gaertner M, Vila M, et al (2015) Ecological impacts of alien species: Quantification, scope, caveats, and recommendations. Bioscience 65:55–63. https://doi.org/10.1093/biosci/biu193

Kupfer JA (2012) Landscape ecology and biogeography: Rethinking landscape metrics

in a post-FRAGSTATS landscape. Prog Phys Geogr 36:400–420.
https://doi.org/10.1177/0309133312439594

Landres PB, Knoght RL, Pickett STA, Cadenasso ML (1998) Ecological effects of administrative boundaries. In: Knight RL, B LP (eds) Stewardship across boundaries. Island Press, Washington D.C., pp 39–64

Laurance WF, Yensen E (1991) Predicting the impacts of edge effects in fragmented habitats. Biol Conserv 55:77–92. https://doi.org/10.1016/0006-3207(91)90006-U

Lausch A, Blaschke T, Haase D, et al (2015) Understanding and quantifying landscape structure - A review on relevant process characteristics, data models and landscape metrics. Ecol Modell 295:31–41.
https://doi.org/10.1016/j.ecolmodel.2014.08.018

Leiva MJ, Fernández-Alés R (2003) Post-dispersive losses of acorns from Mediterranean savannah-like forests and shrublands. For Ecol Manage 176:265–271. https://doi.org/10.1016/S0378-1127(02)00294-3

Levine JM, Vilà M, D'Antonio CM, et al (2003) Mechanisms underlying the impacts of exotic plant invasions. Proc R Soc B Biol Sci 270:775–781.
https://doi.org/10.1098/rspb.2003.2327

Li H, Wu J (2004) Use and misuse of landscape indices. Landsc Ecol 19:389–399.
https://doi.org/10.18773/austprescr.2017.036

Li X, He HS, Bu R, et al (2005) The adequacy of different landscape metrics for various landscape patterns. Pattern Recognit 38:2626–2638.
https://doi.org/10.1016/j.patcog.2005.05.009

Lindenmayer DB, Fischer J (2007) Tackling the habitat fragmentation panchreston.
Trends Ecol Evol 22:127–32. https://doi.org/10.1016/j.tree.2006.11.006

Liu J, Liu X, Wang Y, et al (2020) Landscape composition or configuration: which contributes more to catchment hydrological flows and variations? Landsc Ecol 35:1531–1551. https://doi.org/10.1007/s10980-020-01035-3

Long H, Qu Y (2018) Land use transitions and land management: A mutual feedback perspective. Land use policy 74:111–120.
https://doi.org/10.1016/j.landusepol.2017.03.021

Long JA, Nelson TA, Wulder MA (2010) Characterizing forest fragmentation: Distinguishing change in composition from configuration. Appl Geogr 30:426–435.

https://doi.org/10.1016/j.apgeog.2009.12.002

Machado R, Bayot R, Godinho S, et al (2020) LDTtool: A toolbox to assess landscape dynamics. Environ Model Softw 133:. https://doi.org/10.1016/j.envsoft.2020.104847

Machado R, Godinho S, Pirnat J, et al (2018) Assessment of landscape composition and configuration via spatial metrics combination: conceptual framework proposal and method improvement. Landsc Res 43:652–664. https://doi.org/10.1080/01426397.2017.1336757

Mack RN, Lonsdale WM (2002) Eradicating invasive plants: hard-won lessons for islands. In: Veitch CR, Clout MN (eds) Turning the tide: The Eradication of Invasive Species (Proceedings of the International Conference on Eradication of Island Invasives). pp 164–172

Marchante E, Kjøller A, Struwe S, Freitas H (2008) Invasive Acacia longifolia induce changes in the microbial catabolic diversity of sand dunes. Soil Biol Biochem 40:2563–2568. https://doi.org/10.1016/j.soilbio.2008.06.017

Marchante H, Palhas J, Núñez FAL, Marchante E (2019) Invasive Species Impacts and Management. In: Leal Filho W, Azul A, Brandli L, et al. (eds) Life on Land. Encyclopedia of the UN Sustainable Development Goals. Springer, Cham, pp 1–12

Martins F, Alegria C, Gil A (2016) Mapping invasive alien Acacia dealbata Link using ASTER multispectral imagery: A case study in central-eastern of Portugal. For Syst 25:. https://doi.org/10.5424/fs/2016253-09248

McGarigal K, Cushman SA (2005) The gradient concept of landscape structure. In: Wiens J, Moss M (eds) Issues and Perspectives in Landscape Ecology. Cambridge University Press, Cambridge, pp 112–119

McGarigal K, Cushman SA, 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/f

McGarigal K, Marks BJ (1995) FRAGSTATS: spatial pattern analysis program for quantifying landscape structure

McGarigal K, Tagil S, Cushman SA (2009) Surface metrics: an alternative to patch metrics for the quantification of landscape structure. Landsc Ecol 24:433–450.

https://doi.org/10.1007/s10980-009-9327-y

McRae BH, Shah VB, Mohapatra TK (2013) Circuitscape 4 User Guide. The Nature Conservancy. http://www.circuitscape.org.

Mestre F, Ascensão F, Barbosa AM (2019) gDefrag: A graph-based tool to help defragmenting landscapes divided by linear infrastructures. Ecol Modell 392:1–5. https://doi.org/10.1016/j.ecolmodel.2018.10.012

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: Synthesis. Washington, DC

Minor ES, Gardner RH (2011) Landscape connectivity and seed dispersal characteristics inform the best management strategy for exotic plants. Ecol Appl 21:739–749. https://doi.org/10.1890/10-0321.1

Mitchell M, Powell RA (2003) Linking fitness landscapes with behavior and distribution of animals. In: Bissonette JA, Storch I (eds) Landscape ecology and resource management: linking theory with practice. Island Press, Washington, D.C, pp 93–124

Montello DR (1998) Thinking of scale; The scale of though. Scale Detail Congnition Geogr. Inf.

Montello DR, Golledge RG (1998) Scale and Detail in the Cognition of Geographic Information. Report on a Varenius Specialist Meeting. Santa Barbara, CA

Moody M, Mack R (1988) Controlling the Spread of Plant Invasions : The Importance of Nascent Foci Author (s): Michael E . Moody and Richard N . Mack Published by : British Ecological Society Stable URL : http://www.jstor.org/stable/2403762. J Appl Ecol 25:1009–1021

Moon ZK, Farmer FL (2001) Population Density Surface: A New Approach to an Old Problem. Soc Nat Resour 14:39–51. https://doi.org/10.1080/089419201300199545

Morgado R, Ribeiro PF, Santos JL, et al (2022) Drivers of irrigated olive grove expansion in Mediterranean landscapes and associated biodiversity impacts. Landsc Urban Plan 225:. https://doi.org/10.1016/j.landurbplan.2022.104429

Morgado R, Santana J, Porto M, et al (2020) A Mediterranean silent spring? The effects of olive farming intensification on breeding bird communities. Agric Ecosyst Environ 288:. https://doi.org/10.1016/j.agee.2019.106694

Moser D, Zechmeister HG, Plutzar C, et al (2002) Landscape patch shape complexity as an effective measure for plant species richness in rural landscapes. Landsc Ecol 657–669

Müller F (1998) Gradients in ecological systems. Ecol Modell 108:3–21. https://doi.org/10.1016/S0304-3800(98)00015-5

Muñoz A, Bonal R (2007) Rodents change acorn dispersal behaviour in response to ungulate presence. Oikos 116:1631–1638. https://doi.org/10.1111/j.0030-1299.2007.15710.x

Navarro-Cerrillo RM, Rivas CA, Quinto L, et al (2022) Afforestation on agricultural land in southern Spain : an important driver to improve forest landscape connectivity. New For. https://doi.org/10.1007/s11056-022-09956-4

Ninan KN, Inoue M (2013) Valuing forest ecosystem services: What we know and what we don't. Ecol Econ 93:137–149. https://doi.org/10.1016/j.ecolecon.2013.05.005

Openshaw S (1984) The modifiable areal unit problem. GeoBooks, Norwich, England

Panetta FD (2009) Weed Eradication—An Economic Perspective. Invasive Plant Sci Manag 2:360–368. https://doi.org/10.1614/ipsm-09-003.1

Paracchini ML, Petersen J-E, Hoogeveen Y, et al (2008) High Nature Value Farmland in Europe - An Estimate of the Distribution Patterns on the Basis of Land Cover and Biodiversity Data

Parendes LA, Jones JA (2000) Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conserv Biol 14:64–75. https://doi.org/10.1046/j.1523-1739.2000.99089.x

Parkes J, Panetta F (2009) Eradication of invasive species: progress and emerging issues in the 21st century. In: Clout M, PA W (eds) Invasive species management. A handbook of principles and techniques. Oxford Unviersity Press, Oxford, pp 47–60

Pascual-Hortal L, Saura S (2006) Comparison and development of new graph-based landscape connectivity indices : towards the priorization of habitat patches and corridors for conservation. Landsc Ecol 21:959–967. https://doi.org/10.1007/s10980-006-0013-z

Passos I, Marchante H, Pinho R, Marchante E (2017) What we don't seed: the role of long-lived seed banks as hidden legacies of invasive plants. Plant Ecol 218:1313–

1324. https://doi.org/10.1007/s11258-017-0770-6

Pauchard A, Alaback PB, Edlund EG (2003) Plant invasions in protected areas at multiple scales: Linaria vulgaris (Scrophulariaceae) in the West Yellowstone area. West North Am Nat 63:416–428

Pereira J (2018) Multi-node protection of landscape connectivity: habitat availability and topological reachability. Community Ecol 19:176–185. https://doi.org/10.1556/168.2018.19.2.10

Pinto-Correia T (1993) Threatened landscape in Alentejo , Portugal : the 'montado' and other 'agro-silvo-pastoral' systems. Landsc Urban Plan 24:43–48

Pinto-Correia T, Barroso F, Surová D, Menezes H (2011a) The fuzziness of Montado landscapes: Progress in assessing user preferences through photo-based surveys. Agrofor Syst 82:209–224. https://doi.org/10.1007/s10457-010-9347-2

Pinto-Correia T, Guiomar N, Ferraz-de-Oliveira MI, et al (2018) Progress in Identifying High Nature Value Montados: Impacts of Grazing on Hardwood Rangeland Biodiversity. Rangel Ecol Manag 71:612–625. https://doi.org/10.1016/j.rama.2018.01.004

Pinto-Correia T, Mascarenhas J (1999) Contribution to the extensification/intensification debate: new trends in the Portuguese montado. Landsc Urban Plan 46:125–131

Pinto-Correia T, Ribeiro N, Sá-Sousa P, Sa P (2011b) Introducing the montado, the cork and holm oak agroforestry system of Southern Portugal. Agrofor Syst 82:99–104. https://doi.org/10.1007/s10457-011-9388-1

Plecas M, Gagic V, Jankovic M, et al (2014) Landscape composition and configuration influence cereal aphid – parasitoid – hyperparasitoid interactions and biological control differentially across years. Agric , Ecosyst Environ 183:1–10. https://doi.org/10.1016/j.agee.2013.10.016

Plieninger T (2006) Compatibility of livestock grazing with stand regeneration in Mediterranean holm oak parklands. J Nat Conserv. https://doi.org/10.1016/j.jnc.2005.09.002

Plieninger T, Rolo V, Moreno G (2010) Large-scale patterns of Quercus ilex, Quercus suber, and Quercus pyrenaica regeneration in central-western Spain. Ecosystems 13:644–660. https://doi.org/10.1007/s10021-010-9345-2

Puerta-Piñero C, Pino J, Gómez JM (2012) Direct and indirect landscape effects on Quercus ilex regeneration in heterogeneous environments. Oecologia 170:1009–1020. https://doi.org/10.1007/s00442-012-2373-1

Pulido FJ, Díaz M (2005) Regeneration of a Mediterranean oak: A whole-cycle approach. Écoscience 12:92–102. https://doi.org/10.2980/i1195-6860-12-1-92.1

Pyšek P, Hulme PE, Simberloff D, et al (2020) Scientists' warning on invasive alien species. Biol Rev. https://doi.org/10.1111/brv.12627

R Core Team (2018) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/

Radosevich SR, Stubbs MM, Ghersa CM (2003) Plant invasions — process and patterns. Weed Sci 51:254–259

Rempel RS, Kaukinen D, Carr AP (2012) Patch Analyst and Patch Grid. Ontario Ministry of Natural Resources. Centre for Northern Forest Ecosystem Research, Thunder Bay, Ontario. Available from: http://www.cnfer.on.ca/SEP/

Rodriguez-Galiano V, Chica-Olmo M (2012) Land cover change analysis of a Mediterranean area in Spain using different sources of data: Multi-seasonal Landsat images, land surface temperature, digital terrain models and texture. Appl Geogr 35:208–218. https://doi.org/10.1016/j.apgeog.2012.06.014

Rosenweig ML (1995) Species diversity in space and time. Cambridge University Press, Cambridge

Roura-Pascual N, Krug RM, Richardson DM, Hui C (2010) Spatially-explicit sensitivity analysis for conservation management: Exploring the influence of decisions in invasive alien plant management. Divers Distrib 16:426–438. https://doi.org/10.1111/j.1472-4642.2010.00659.x

Saunders DA, Hobbs RJ, Margules CR (1991) Biological Consequences of Ecosystem Fragmentation : A Review. Conserv Biol 5:18–32

Saura S, Rubio L (2010) A common currency for the different ways in which patches and links can contribute to habitat availability and connectivity in the landscape. Ecography (Cop) 33:523–537. https://doi.org/10.1111/j.1600-0587.2009.05760.x

Saura S, Torné J (2009) Conefor Sensinode 2.2: A software package for quantifying the importance of habitat patches for landscape connectivity. Environ Model Softw

24:135–139. https://doi.org/10.1016/j.envsoft.2008.05.005

Scherreiks P, Gossner MM, Ambarlı D, et al (2022) Present and historical landscape structure shapes current species richness in Central European grasslands. Landsc Ecol 7:745–762. https://doi.org/10.1007/s10980-021-01392-7

Schindler S, Staska B, Adam M, et al (2015) Alien species and public health impacts in Europe: A literature review. NeoBiota 27:1–23. https://doi.org/10.3897/neobiota.27.5007

Silva LP, Mata VA (2019) Stop harvesting olives at night — it kills millions of songbirds. Nature 569:

Silveira A, Ferrão J, Muñoz-Rojas J, et al (2018) The sustainability of agricultural intensification in the early st insights from the olive oil production in Alentejo (Southern Portugal). In: Delicado A, Domingos N, de Sousa L (eds) Changing Societies: Legacies and Challenges. Vol. III. The Diverse Worlds of Sustainability. Imprensa de Ciências Sociais, Lisbon, pp 247–275

Simberloff D, Martin JL, Genovesi P, et al (2013) Impacts of biological invasions: What's what and the way forward. Trends Ecol Evol 28:58–66. https://doi.org/10.1016/j.tree.2012.07.013

Simoncini R, Ring I, Sandström C, et al (2019) Constraints and opportunities for mainstreaming biodiversity and ecosystem services in the EU's Common Agricultural Policy: Insights from the IPBES assessment for Europe and Central Asia. Land use policy 88:104099. https://doi.org/10.1016/j.landusepol.2019.104099

Škornik S, Lončar M, Kaligarič M (2006) Vegetation of Silicicolous Grasslands of the Highlands of North-Eastern Slovenia. Hacquetia 5:193–211

St-Laurent M-H, Dussault C, Ferron J, Gagnon R (2009) Dissecting habitat loss and fragmentation effects following logging in boreal forest: Conservation perspectives from landscape simulations. Biol Conserv 142:2240–2249. https://doi.org/10.1016/j.biocon.2009.04.025

Steiniger S, Hay GJ (2009) Free and open source geographic information tools for landscape ecology. Ecol Inform 4:183–195. https://doi.org/10.1016/j.ecoinf.2009.07.004

Steiniger S, Hunter AJS (2013) The 2012 free and open source GIS software map - A

guide to facilitate research, development, and adoption. Comput Environ Urban Syst 39:136–150. https://doi.org/10.1016/j.compenvurbsys.2012.10.003

Stoate C, Báldi A, Beja P, et al (2009) Ecological impacts of early 21st century agricultural change in Europe - A review. J Environ Manage 91:22–46. https://doi.org/10.1016/j.jenvman.2009.07.005

Stoica IV, Zamfir D, Vîrghileanu M (2021) Evaluating the territorial impact of built-up area expansion in the surroundings of bucharest (Romania) through a multilevel approach based on landsat satellite imagery. Remote Sens 13:. https://doi.org/10.3390/rs13193969

Šumrada T, Japelj A, Verbič M, Erjavec E (2022) Farmers' preferences for result-based schemes for grassland conservation in Slovenia. J Nat Conserv 126143. https://doi.org/10.1016/j.jnc.2022.126143

Šumrada T, Kmecl P, Erjavec E (2021a) Do the EU's Common agricultural policy funds negatively affect the diversity of farmland birds? Evidence from Slovenia. Agric Ecosyst Environ 306:. https://doi.org/10.1016/j.agee.2020.107200

Šumrada T, Vreš B, Čelik T, et al (2021b) Are result-based schemes a superior approach to the conservation of High Nature Value grasslands? Evidence from Slovenia. Land use policy. https://doi.org/10.1016/j.landusepol.2021.105749

Swift A, Liu L, Uber J (2008) Reducing MAUP bias of correlation statistics between water quality and GI illness. Comput Environ Urban Syst 32:134–148. https://doi.org/10.1016/j.compenvurbsys.2008.01.002

Syrbe R, Walz U (2012) Spatial indicators for the assessment of ecosystem services : Providing , benefiting and connecting areas and landscape metrics. Ecol Indic 21:80–88. https://doi.org/10.1016/j.ecolind.2012.02.013

Tischendorf L (2001) Can landscape indices predict ecological processes consistently? Landsc Ecol 16:235–254. https://doi.org/10.1023/A:1011112719782

Tome D, Denac D, Vrezec A (2020) Mowing is the greatest threat to Whinchat Saxicola rubetra nests even when compared to several natural induced threats. J Nat Conserv 54:125781. https://doi.org/10.1016/j.jnc.2019.125781

Turner MG (1989) Landscape Ecology : The Effect of Pattern on Process Published by : Annual Reviews Linked references are available on JSTOR for this article : LANDSCAPE ECOLOGY : The Effect of Pattern on Process '. Annu Rev Ecol Syst

20:171–197

Turner MG (2005) Landscape ecology: What is the state of the science? Annu Rev Ecol Evol Syst 36:319–344. https://doi.org/10.1146/annurev.ecolsys.36.102003.152614

Turner MG (1990) Spatial and temporal analysis of landscape patterns. Landsc Ecol 4:21–30

Turner MG, Gardner RH (1991) Quantitative Methods in Landscape Ecology: The Analysis and Interpretation of Landscape Heterogeneity. Springer, New York

Uuemaa E, Antrop M, Roosaare J, et al (2009) Landscape metrics and indices: An overview of their use in landscape research. Living Rev Landsc Res 3:1–28. https://doi.org/10.12942/lrlr-2009-1

Uuemaa E, Mander Ü, Marja R (2013) Trends in the use of landscape spatial metrics as landscape indicators: A review. Ecol Indic 28:100–106. https://doi.org/10.1016/j.ecolind.2012.07.018

Van Doorn AM, Pinto-Correia T (2007) Differences in land cover interpretation in landscapes rich in cover gradients: Reflections based on the montado of South Portugal. Agrofor Syst 70:169–183. https://doi.org/10.1007/s10457-007-9055-8

Van Kleunen M, Weber E, Fischer M (2010) A meta-analysis of trait differences between invasive and non-invasive plant species. Ecol Lett 13:235–245. https://doi.org/10.1111/j.1461-0248.2009.01418.x

van Vliet J, de Groot HLF, Rietveld P, Verburg PH (2015) Manifestations and underlying drivers of agricultural land use change in Europe. Landsc Urban Plan 133:24–36. https://doi.org/10.1016/j.landurbplan.2014.09.001

Van Vooren L, Reubens B, Broekx S, et al (2018) Assessing the impact of grassland management extensification in temperate areas on multiple ecosystem services and biodiversity. Agric Ecosyst Environ 267:201–212. https://doi.org/10.1016/j.agee.2018.08.016

van Wilgen BW, Forsyth GG, Le Maitre DC, et al (2012) An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. Biol Conserv 148:28–38. https://doi.org/10.1016/j.biocon.2011.12.035

Veen P, Jefferson R, de Smidt J, van der Straaten J (2009) Grasslands in Europe of high

nature value by Veen , P ., Jefferson , R ., de Smidt , J ., Book Review Grasslands in Europe of high nature value . KNNV Publishing, Zeist

Vilà M, Espinar JL, Hejda M, et al (2011) Ecological impacts of invasive alien plants: A meta-analysis of their effects on species, communities and ecosystems. Ecol Lett 14:702–708. https://doi.org/10.1111/j.1461-0248.2011.01628.x

Vosse S, Esler KJ, Richardson DM, Holmes PM (2008) Can riparian seed banks initiate restoration after alien plant invasion? Evidence from the Western Cape, South Africa. South African J Bot 74:432–444. https://doi.org/10.1016/j.sajb.2008.01.170

Waller LA, Gotway CA (2004) Applied Spatial Statistics for Public Health Data. John Wiley and Sons, New York

WallisDeVries MF, Poschlod P, Willems JH (2002) Challenges for the conservation of calcareous grasslands in northwestern Europe: Integrating the requirements of flora and fauna. Biol Conserv 104:265–273. https://doi.org/10.1016/S0006-3207(01)00191-4

Wang X, Cumming SG (2009) Modeling configuration dynamics of harvested forest landscapes in the Canadian boreal plains. Landsc Ecol 24:229–241. https://doi.org/10.1007/s10980-008-9301-0

Ward M, Saura S, Williams B, et al (2020) Only ten percent of the global terrestrial protected area network is connected via intact land. Nat Commun 1–10. https://doi.org/10.1101/2020.01.28.920488

Wiens JA (1996) Wildlife in patchy environments: Metapopulations, mosaics and management. In: McCullough DR (ed) Metapopulations and wildlife conservation. Island Press, Washington, DC, pp 53–84

Wilcove DS, McLellan CH, Dobson AP (1986) Habitat fragmentation in the temperate. In: Soulé ME (ed) Conservation Biology. Sinauer Associates, Sunderland, MA, pp 237–256

With KA (1997) The Application of Neutral Landscape Models in Conservation Biology. Conserv Biol 11:1069–1080

With KA (2019) Essentials of landscape ecology. Oxford University Press, Oxford

Zaccarelli N, Petrosillo I, Zurlini G, Riitters KH (2008) Source/sink patterns of disturbance and cross-scale mismatches in a panarchy of social-ecological

landscapes. Ecol Soc 13:. https://doi.org/10.5751/ES-02416-130126

Zaragozí B, Belda A, Linares J, et al (2012) A free and open source programming library
for landscape metrics calculations. Environ Model Softw 31:131–140.
https://doi.org/10.1016/j.envsoft.2011.10.009

APPENDICES

Appendix I – LDT dichotomous key

Appendix II – Example of LDTtool in ArcGIS model builder

Appendix III – Example of LDT4QGIS script

Appendix IV – Published articles

APPENDIX I – LDT dichotomous key
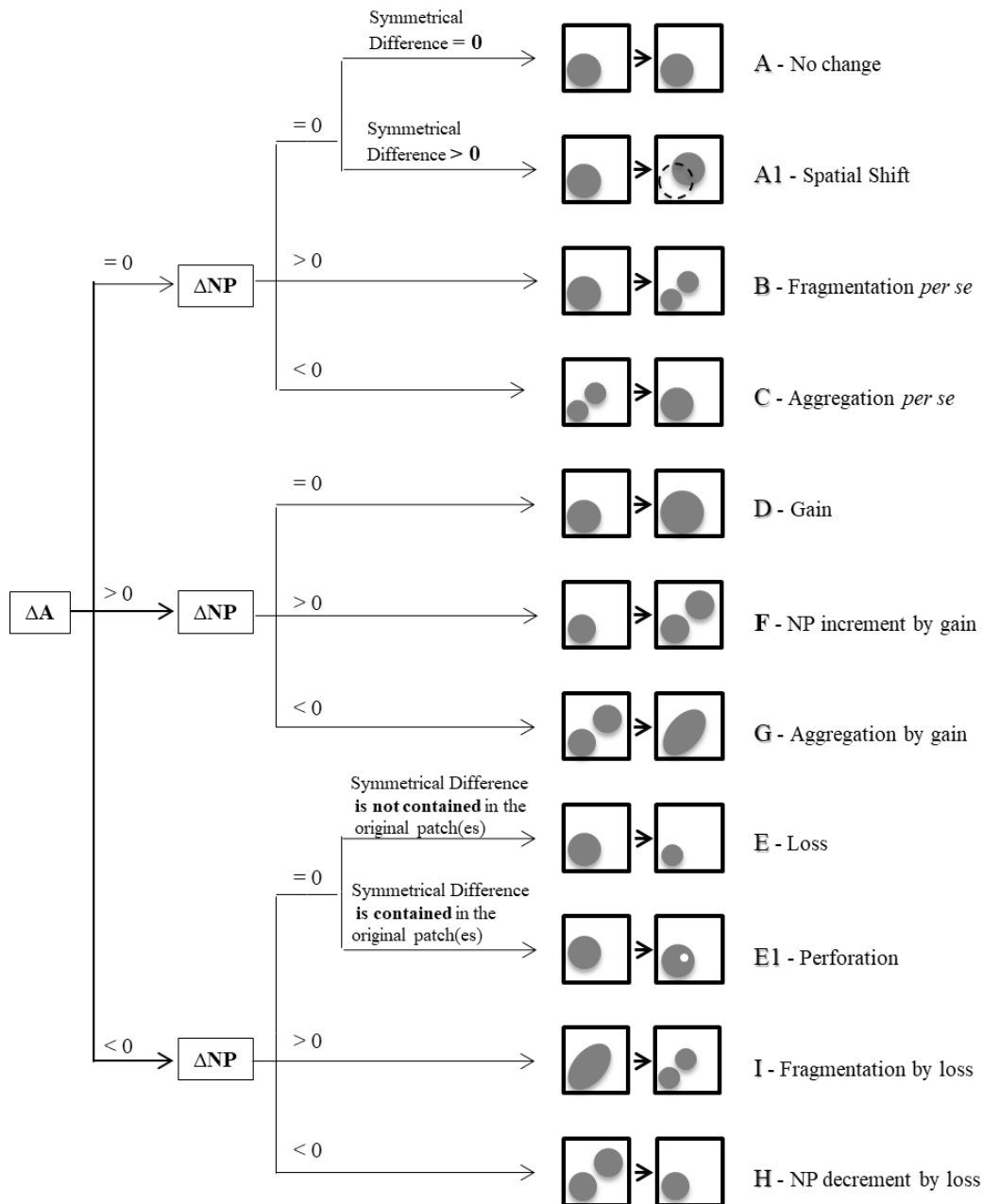
(A) If there is no change in an area or in the number of patches (NP) we assume that landscape did not change; (A1) If there is no change in area and NP but the patch(es) is/are not located on the same place as before, a spatial shift has occurred (B) If the area remained the same but the NP increased, it means a fragmentation occurred; (C) If the area remained the same but NP decreased, then an aggregation took place; (D) If the area increased and the NP is equal, it represents a gain of area; (E) If the area decreased and the NP did not change, there is a loss of area; (E1) Particular case of the previous ToD in which the loss amount created a clearing; (F) If both area and NP increased, it led to new patch creation; (G) If the area increased and NP decreased, an aggregation occurred due to area gain; (H) If both area and NP decreased, a patch decrement occurred due to area loss; (I) If area decreased and the NP increased, it means that fragmentation occurred due to area loss.

LDTtool: Tool 1.2.1 - Landscape Dynamics Types 3M Squares (with perforation)
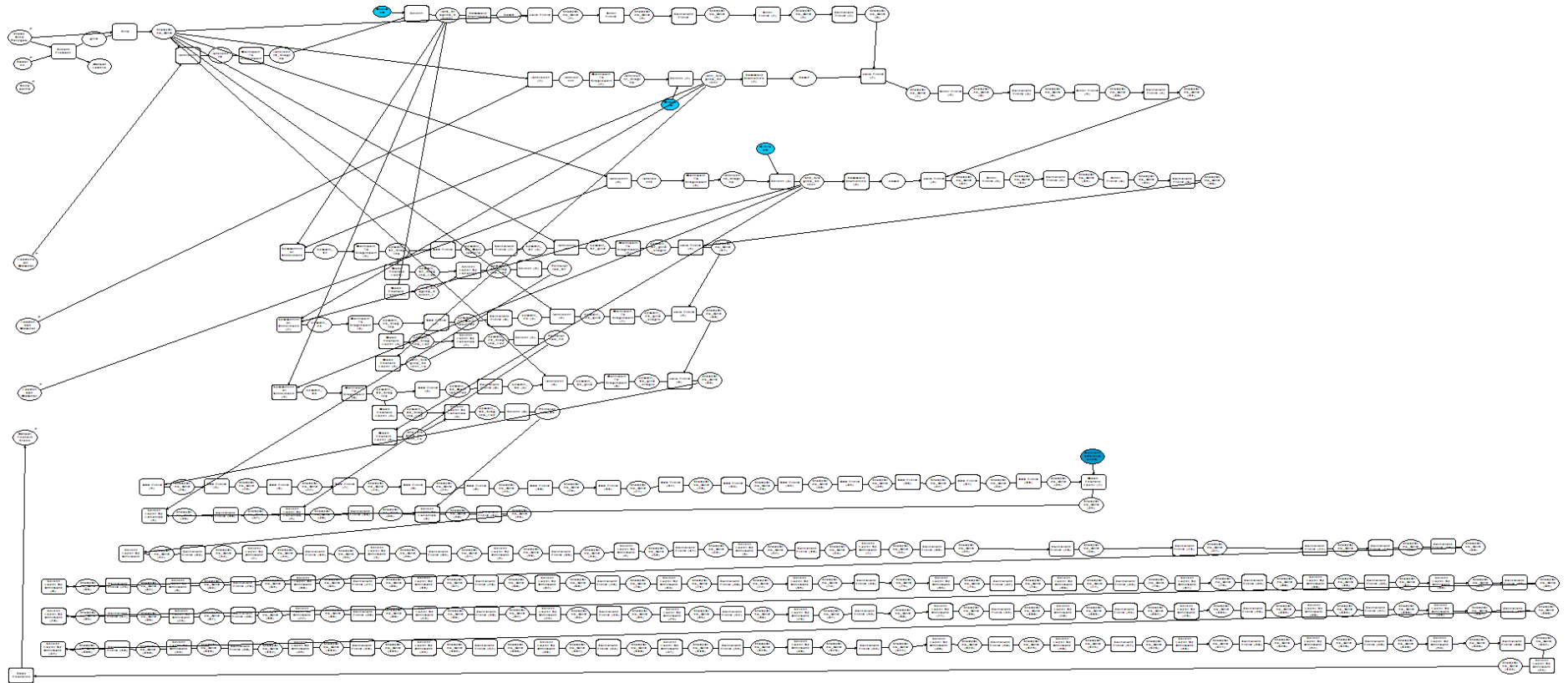
Python code

```python
# -*- coding: utf-8 -*-

import os, sys
import ogr
from math import ceil
from qgis.PyQt.QtCore import QCoreApplication
from qgis.core import (QgsProcessing,
            QgsFeatureSink,
            QgsProcessingAlgorithm,
            QgsProcessingParameterNumber,
            QgsProcessingParameterFeatureSink,
            QgsProcessingParameterVectorLayer,
            QgsProcessingParameterBoolean,
            QgsProcessingParameterEnum,
            QgsVectorLayer,
            QgsProject,
            QgsCoordinateReferenceSystem,
            QgsFeatureRequest,
            QgsProcessingUtils,
            QgsProcessingContext)
from qgis import processing
from qgis.utils import iface
from processing.core.ProcessingConfig import ProcessingConfig
from processing.tools import dataobjects


class LDT4QGIS(QgsProcessingAlgorithm):
    # Constants used to refer to parameters and outputs
    MOMENT = 'Moments'
    AREATYPE = 'AreaType'
    INPUT_SA = 'Study Polygon Area'
    INPUT_LM1 = 'Landscape Moment 1'
    INPUT_LM2 = 'Landscape Moment 2'
    INPUT_LM3 = 'Landscape Moment 3'
    INPUT_PATCH = 'Keep patches equal or larger than (sq. meters)'
    INPUT_SQDIM = 'Squares width and height (meters)'
    OUTPUT = 'Output Shapefile'
    PERFORATION = 'Perforation'
    m3 = False
    dstr = False

    def tr(self, string):
        return QCoreApplication.translate('Processing', string)

    def createInstance(self):
        return LDT4QGIS()

    def name(self):
        return 'LDT4QGIS'

    def displayName(self):
        return self.tr('Landscape Dynamics Types')

    def group(self):
        return self.tr('LDT4QGIS')

    def groupId(self):
        return 'LDT4QGIS'

    def shortHelpString(self):

        return self.tr("""Calculates Landscape Dynamic Types for 2 or 3 moments, using squares or districts as analytical units.\n
        Study Polygon Area:   Study area boundary\n
        Landscape Moment 1:   Landscape (LULC category) shapefile in the earliest moment of analysis\n
        Landscape Moment 2:   Landscape (LULC category) shapefile in the second moment of analysis\n
```

Landscape Moment 3:   Landscape (LULC category) shapefile in the latest moment of analysis (only valid for 3 moments analysis)\n
    Keep patches equal or larger than (sq. meters):   Minimum patch size in square meters.(All the patches smaller than this threshold value will be discarded from the analysis)\n
    Squares width and height (meters):   Insert square size, width and height, in meters (only valid for squares as analytical unit)\n
    Output Shapefile: Select the name of the output shapefile\n""")

```python
 def initAlgorithm(self, config=None):

    # Parameters
    self.addParameter(
        QgsProcessingParameterEnum(
            self.MOMENT,
            self.tr('Moments'),
            ["2", "3"]
        )

    )
    self.addParameter(
        QgsProcessingParameterEnum(
            self.AREATYPE,
            self.tr('Type of Analysis'),
            ["Squares", "Districts"]
        )

    )
    self.addParameter(
        QgsProcessingParameterVectorLayer(
            self.INPUT_SA,
            self.tr('Study Area Polygon'),
            [QgsProcessing.TypeVectorPolygon]
        )
    )
    self.addParameter(
        QgsProcessingParameterVectorLayer(
            self.INPUT_LM1,
            self.tr('Landscape Moment 1'),
            [QgsProcessing.TypeVectorPolygon]
        )
    )
    self.addParameter(
        QgsProcessingParameterVectorLayer(
            self.INPUT_LM2,
            self.tr('Landscape Moment 2'),
            [QgsProcessing.TypeVectorPolygon]
        )
    )
    self.addParameter(
        QgsProcessingParameterVectorLayer(
            self.INPUT_LM3,
            self.tr('Landscape Moment 3 (Ignore if 2 Moment analysis is selected)'),
            [QgsProcessing.TypeVectorPolygon],
            optional = True
        )
    )
    self.addParameter(
        QgsProcessingParameterNumber(
            self.INPUT_PATCH,
            self.tr('Keep patches equal or larger than (sq. meters)'),
        )
    )
    self.addParameter(
        QgsProcessingParameterNumber(
            self.INPUT_SQDIM,
            self.tr('Squares width and height (meters) (Ignore if Districts analysis is selected)'),
            optional = True
        )
    )
    self.addParameter(
        QgsProcessingParameterBoolean(
            self.PERFORATION,
```

```python
            self.tr('Perforation'),
            defaultValue = False
        )
    )
    self.addParameter(
        QgsProcessingParameterFeatureSink(
            self.OUTPUT,
            self.tr('Output Shapefile')
        )
    )

def processAlgorithm(self, parameters, context, feedback):

    QgsProcessingContext().setInvalidGeometryCheck(QgsFeatureRequest.GeometrySkipInvalid)

    # Input and Output variables
    mmt = self.parameterAsString(
        parameters,
        self.MOMENT,
        context
    )
    atyp = self.parameterAsString(
        parameters,
        self.AREATYPE,
        context
    )
    study_areaS = self.parameterAsCompatibleSourceLayerPath(
        parameters,
        self.INPUT_SA,
        context,
        ['shp']
    )
    land1S = self.parameterAsCompatibleSourceLayerPath(
        parameters,
        self.INPUT_LM1,
        context,
        ['shp']
    )
    land2S = self.parameterAsCompatibleSourceLayerPath(
        parameters,
        self.INPUT_LM2,
        context,
        ['shp']
    )
    land3S = self.parameterAsCompatibleSourceLayerPath(
        parameters,
        self.INPUT_LM3,
        context,
        ['shp']
    )
    min_size = self.parameterAsInt(
        parameters,
        self.INPUT_PATCH,
        context
    )
    squares = self.parameterAsInt(
        parameters,
        self.INPUT_SQDIM,
        context
    )
    pref = self.parameterAsBool(
        parameters,
        self.PERFORATION,
        context
    )
    dest_id_name = self.parameterAsFileOutput(
        parameters,
        self.OUTPUT,
        context,
    )
```

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```python
# configCheck
#m3 = False
if mmt == "1":
    self.m3 = True
#dstr = False
if atyp == "1":
    self.dstr = True


# Temporary Folder
dirTemp = os.path.join(os.path.dirname(dest_id_name), "tempFolderLDT")
if not os.path.exists(dirTemp):
    os.makedirs(dirTemp)
else:
    for root, dirs, files in os.walk(dirTemp):
        for file in files:
            path = os.path.join(dirTemp, file)
            os.remove(path)
    feedback.pushInfo("Temporary folder already exists")

#Buffer 0
inlyrSD = QgsVectorLayer(study_areaS, 'temp', 'ogr')
inlyrL1 = QgsVectorLayer(land1S, 'temp', 'ogr')
inlyrL2 = QgsVectorLayer(land2S, 'temp', 'ogr')

study_areaS = os.path.join(dirTemp, os.path.basename(study_areaS))
land1S = os.path.join(dirTemp, os.path.basename(land1S))
land2S = os.path.join(dirTemp, os.path.basename(land2S))

processing.run("qgis:buffer", {"INPUT": inlyrSD, "DISTANCE": 0, "OUTPUT": study_areaS})
processing.run("qgis:buffer", {"INPUT": inlyrL1, "DISTANCE": 0, "OUTPUT": land1S})
processing.run("qgis:buffer", {"INPUT": inlyrL2, "DISTANCE": 0, "OUTPUT": land2S})

del inlyrSD, inlyrL1, inlyrL2

#feedback.pushInfo(m3)
if self.m3:
    inlyrL3 = QgsVectorLayer(land3S, 'temp', 'ogr')
    land3S = os.path.join(dirTemp, os.path.basename(land3S))
    processing.run("qgis:buffer", {"INPUT": inlyrL3, "DISTANCE": 0, "OUTPUT": land3S})
    del inlyrL3

# Fields
Fid_Field = "FID"
areaInicial_field = "Area"
freq_field_1 = "NP1"
freq_field_2 = "NP2"
area_1 = "area_1"
area_2 = "area_2"
presence_1 = "Presence1"
presence_2 = "Presence2"
var_area_12 = "varArea_12"
var_NP_12 = "var_NP_12"
ToD_12 = "ToD_12"
sym12 = "symdif_12"
perforation12 = "Perf_12"

if self.m3:
    freq_field_3 = "NP3"
    area_3 = "area_3"
    presence_3 = "Presence3"
    var_area_23 = "varArea_23"
    var_area_13 = "varArea_13"
    var_NP_23 = "var_NP_23"
    var_NP_13 = "var_NP_13"
    ToD_23 = "ToD_23"
    ToD_13 = "ToD_13"
    sym23 = "symdif_23"
    sym13 = "symdif_13"
    perforation23 = "Perf_23"
    perforation13 = "Perf_13"
```

```python
# Temporary Shapefiles
fishnet_temp1 = os.path.join(dirTemp, "fishnet_t1.shp")
fishnet_temp2 = os.path.join(dirTemp, "fishnet_t2.shp")
intersect_temp1 = os.path.join(dirTemp, "intersect_t1.shp")
intersect_temp2 = os.path.join(dirTemp, "intersect_t2.shp")
multisingle_temp1 = os.path.join(dirTemp, "multisingle_t1.shp")
multisingle_temp2 = os.path.join(dirTemp, "multisingle_t2.shp")
select_temp1 = os.path.join(dirTemp, "select_t1.shp")
select_temp2 = os.path.join(dirTemp, "select_t2.shp")
symdif12 = os.path.join(dirTemp, "symdif12.shp")
ms_symdif12 = os.path.join(dirTemp, "mu_symdif12.shp")
#symdif12_fix = os.path.join(dirTemp, "symdif12_fix.shp")
pref12 = os.path.join(dirTemp, "pref12.shp")
prefBuff12 = os.path.join(dirTemp, "prefBuff12.shp")
fishnet_temp3 = os.path.join(dirTemp, "fishnet_t3.shp")

if self.m3:
    intersect_temp3 = os.path.join(dirTemp, "intersect_t3.shp")
    multisingle_temp3 = os.path.join(dirTemp, "multisingle_t3.shp")
    select_temp3 = os.path.join(dirTemp, "select_t3.shp")
    symdif23 = os.path.join(dirTemp, "symdif23.shp")
    pref23 = os.path.join(dirTemp, "pref23.shp")
    prefBuff23 = os.path.join(dirTemp, "prefBuff23.shp")
    ms_symdif23 = os.path.join(dirTemp, "mu_symdif23.shp")
    symdif13 = os.path.join(dirTemp, "symdif13.shp")
    pref13 = os.path.join(dirTemp, "pref13.shp")
    prefBuff13 = os.path.join(dirTemp, "prefBuff13.shp")
    ms_symdif13 = os.path.join(dirTemp, "mu_symdif13.shp")
    fishnet_temp4 = os.path.join(dirTemp, "fishnet_t4.shp")
    fishnet_temp5 = os.path.join(dirTemp, "fishnet_5.shp")


# Functions
def create_fishnet(inputSRC,outputGridfn,gridHeight,gridWidth):
    inDriver = ogr.GetDriverByName("ESRI Shapefile")
    inDataSource = inDriver.Open(inputSRC, 0)
    inLayer = inDataSource.GetLayer()
    extent = inLayer.GetExtent()
    srsSource = inLayer.GetSpatialRef()

    xmin = float(extent[0])
    xmax = float(extent[1])
    ymin = float(extent[2])
    ymax = float(extent[3])
    gridWidth = float(gridWidth)
    gridHeight = float(gridHeight)

    rows = ceil((ymax-ymin)/gridHeight)
    cols = ceil((xmax-xmin)/gridWidth)

    ringXleftOrigin = xmin
    ringXrightOrigin = xmin + gridWidth
    ringYtopOrigin = ymin
    ringYbottomOrigin = ymin+gridHeight

    outDriver = ogr.GetDriverByName('ESRI Shapefile')
    if os.path.exists(outputGridfn):
        os.remove(outputGridfn)
    outDataSource = outDriver.CreateDataSource(outputGridfn)
    outLayer = outDataSource.CreateLayer(outputGridfn,srs=srsSource,geom_type=ogr.wkbPolygon)
    featureDefn = outLayer.GetLayerDefn()

    countcols = 0
    while countcols < cols:
        countcols += 1

        ringYtop = ringYtopOrigin
        ringYbottom =ringYbottomOrigin
        countrows = 0
```

```
        while countrows < rows:
            countrows += 1
            ring = ogr.Geometry(ogr.wkbLinearRing)
            ring.AddPoint(ringXleftOrigin, ringYbottom)
            ring.AddPoint(ringXrightOrigin, ringYbottom)
            ring.AddPoint(ringXrightOrigin, ringYtop)
            ring.AddPoint(ringXleftOrigin, ringYtop)
            ring.AddPoint(ringXleftOrigin, ringYbottom)
            poly = ogr.Geometry(ogr.wkbPolygon)
            poly.AddGeometry(ring)

            outFeature = ogr.Feature(featureDefn)
            outFeature.SetGeometry(poly)
            outLayer.CreateFeature(outFeature)
            outFeature = None

            ringYtop = ringYtop + gridHeight
            ringYbottom = ringYbottom + gridHeight

        ringXleftOrigin = ringXleftOrigin + gridWidth
        ringXrightOrigin = ringXrightOrigin + gridWidth

    outDataSource = None

    layer = QgsVectorLayer(outputGridfn, 'temp', 'ogr')
    lyrCRS = layer.crs().authid()

    return(outputGridfn, lyrCRS)

def clip_shp(sourceFeature, clipFeature, outputFeature):
    inlyr_src = QgsVectorLayer(sourceFeature, 'temp', 'ogr')
    inlyr_clip = QgsVectorLayer(clipFeature, 'temp', 'ogr')
    processing.run("qgis:clip", {"INPUT": inlyr_src, "OVERLAY": inlyr_clip, "OUTPUT": outputFeature})

    del inlyr_src, inlyr_clip
    return(outputFeature)

def intersect_simple(input_shp, overlay, output_shp):
    inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')
    processing.run("qgis:intersection", {"INPUT": input_shp, "OVERLAY": overlay, "OUTPUT": output_shp})
    del inlyr
    return(output_shp)

def multi_to_single(input_shp, output_shp):
    inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')
    processing.run("qgis:multiparttosingleparts", {"INPUT": input_shp, "OUTPUT": output_shp})
    del inlyr
    return(output_shp)

def update_field_area(input_shp, field_area):

    driver = ogr.GetDriverByName("ESRI Shapefile")
    dataSource = driver.Open(input_shp, 1) # 0 means read-only. 1 means writeable.
    layer = dataSource.GetLayer()

    new_field = ogr.FieldDefn(field_area, ogr.OFTReal)
    new_field.SetPrecision(2)

    if layer.GetLayerDefn().GetFieldIndex(field_area) >= 0:
        for feature in layer:
            geom = feature.GetGeometryRef()
            area = geom.GetArea()
            feature.SetField(field_area, area)
            layer.SetFeature(feature)
        dataSource = None
    else:
        layer.CreateField(new_field)
        for feature in layer:
            geom = feature.GetGeometryRef()
            area = geom.GetArea()
            feature.SetField(field_area, area)
```

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```python
        layer.SetFeature(feature)
    dataSource = None

def update_field_fid(input_shp, field_fid):

    driver = ogr.GetDriverByName("ESRI Shapefile")
    dataSource = driver.Open(input_shp, 1) # 0 means read-only. 1 means writeable.
    layer = dataSource.GetLayer()

    fc = layer.GetFeatureCount()
    count = 1
    for feature in layer:
        feature.SetField(field_fid, count)
        layer.SetFeature(feature)
        count+=1
    dataSource = None

def select_att(input_shp, field, operator, value, output_shp):

    select = ("{field1}{operator1}{value1}").format(
        field1 = field,
        operator1 = operator,
        value1 = value
    )

    inds = ogr.Open(input_shp)
    inlyr=inds.GetLayer()
    inlyr.SetAttributeFilter(select)
    driver = ogr.GetDriverByName('ESRI Shapefile')
    outds = driver.CreateDataSource(output_shp)
    outlyr = outds.CopyLayer(inlyr, output_shp)
    inds = None
    del inlyr,inds,outlyr,outds

    return(output_shp)

def stats_by_AreaFID(input_shp, field_AREA, field_FID, field_AreaFinal, field_CountFinal):

    fidUnique = []
    sumA = []
    countA = []

    inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')
    fc = inlyr.featureCount()

    list_f = []
    for i in range(0, fc):
        feat = inlyr.getFeature(i)
        fid_value = feat[field_FID]
        list_f.append(fid_value)
    set_fid = set(list_f)

    for f in set_fid:
        fidUnique.append(f)
        select = ('"{field1}" {operator1} {value1}').format(
            field1 = field_FID,
            operator1 = '=',
            value1 = f
        )
        inlyr.selectByExpression(select)
        count = inlyr.selectedFeatureCount()
        countA.append(count)

        ids = inlyr.selectedFeatureIds()
        request = QgsFeatureRequest()
        request.setFilterFids(ids)
        temp_a = []
        for feature in inlyr.getFeatures(request):
            reg = feature[field_AREA]
            temp_a.append(reg)
            del reg
```

```python
        del ids, request, count
        sumA.append(sum(temp_a))

    dict_fid = {field_FID: fidUnique, field_AreaFinal: sumA, field_CountFinal: countA}
    inlyr = None
    del inlyr
    return(dict_fid)

def create_field(input_shp, field_name, field_type, field_precision, field_width):

    if field_type == 0:
        field_type = ogr.OFTInteger
        new_field = ogr.FieldDefn(field_name, field_type)
        new_field.SetWidth(field_width)
    elif field_type == 1:
        field_type = ogr.OFTReal
        new_field = ogr.FieldDefn(field_name, field_type)
        new_field.SetWidth(field_width)
        new_field.SetPrecision(field_precision)
    elif field_type == 2:
        field_type = ogr.OFTString
        new_field = ogr.FieldDefn(field_name, field_type)
        new_field.SetWidth(field_width)

    driver = ogr.GetDriverByName("ESRI Shapefile")
    dataSource = driver.Open(input_shp, 1) # 0 means read-only. 1 means writeable.
    layer = dataSource.GetLayer()

    if layer.GetLayerDefn().GetFieldIndex(field_name) >= 0:
        dataSource = None
    else:
        layer.CreateField(new_field)
        dataSource = None

def join_FID_AREA(input_shp, input_dict, join_field, field_COUNT, field_AREA):

    inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')

    fc = len(input_dict[join_field])

    for f in range(0, fc):

        COUNT_value = input_dict[field_COUNT][f]
        AREA_value = input_dict[field_AREA][f]
        select = ('"{field1}" {operator1} {value1}').format(
            field1 = join_field,
            operator1 = '=',
            value1 = input_dict[join_field][f]
        )
        inlyr.selectByExpression(select)

        # get the list of selected ids
        ids = inlyr.selectedFeatureIds()

        # create the request with the selected ids
        request = QgsFeatureRequest()
        request.setFilterFids(ids)

        for feature in inlyr.getFeatures(request):
            inlyr.startEditing()
            feature.setAttribute(feature.fieldNameIndex(field_COUNT), COUNT_value)
            feature.setAttribute(feature.fieldNameIndex(field_AREA), AREA_value)
            inlyr.updateFeature(feature)
        inlyr.commitChanges()
        iface.vectorLayerTools().stopEditing(inlyr)

    #null to 0
    inlyr.selectByExpression(('"{field1}" is NULL').format(field1 = field_COUNT))
    ids = inlyr.selectedFeatureIds()
    request = QgsFeatureRequest()
    request.setFilterFids(ids)
```

```python
    for feature in inlyr.getFeatures(request):
        inlyr.startEditing()
        feature.setAttribute(feature.fieldNameIndex(field_COUNT), 0)
        feature.setAttribute(feature.fieldNameIndex(field_AREA), 0)
        inlyr.updateFeature(feature)
    inlyr.commitChanges()
    iface.vectorLayerTools().stopEditing(inlyr)
    inlyr = None

def calcField(input_shp, target_field, target_value, fieldNULL, fieldCalc):

    inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')

    inlyr.startEditing()
    for feature in inlyr.getFeatures():
        #inlyr.startEditing()
        #feature.setAttribute(feature.fieldNameIndex(target_field), target_value)
        #inlyr.updateFeature(feature)
        feature[target_field] = target_value

        pos = feature[fieldNULL]
        if not pos:
            feature[fieldNULL] = feature[fieldCalc]
            #feature.setAttribute(feature.fieldNameIndex(fieldNULL), feature.attributes()[feature.fieldNameIndex(fieldCalc)] )

        inlyr.updateFeature(feature)

    inlyr.commitChanges()
    iface.vectorLayerTools().stopEditing(inlyr)
    inlyr = None

def calcField_byLocation(in_layer, intersect, target_field, target_value):
    inlyr = QgsVectorLayer(in_layer, 'temp', 'ogr')
    inlyrI = QgsVectorLayer(intersect, 'temp', 'ogr')
    processing.run('qgis:selectbylocation', {"INPUT":inlyr, "PREDICATE": 1, "INTERSECT": inlyrI, "METHOD": 0})

    ids = inlyr.selectedFeatureIds()
    request = QgsFeatureRequest()
    request.setFilterFids(ids)

    for feature in inlyr.getFeatures(request):
        inlyr.startEditing()
        feature.setAttribute(feature.fieldNameIndex(target_field), target_value)
        inlyr.updateFeature(feature)

    inlyr.commitChanges()
    iface.vectorLayerTools().stopEditing(inlyr)
    inlyr = None

def calcField_bySelection_single(input_shp, select_field, select_operator, select_value, target_field, target_value):

    inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')

    select = ('"{field1}" {operator1} {value1}').format(
        field1 = select_field,
        operator1 = select_operator,
        value1 = select_value
    )
    inlyr.selectByExpression(select)

    ids = inlyr.selectedFeatureIds()
    request = QgsFeatureRequest()
    request.setFilterFids(ids)

    for feature in inlyr.getFeatures(request):
        inlyr.startEditing()
        feature.setAttribute(feature.fieldNameIndex(target_field), target_value)
        inlyr.updateFeature(feature)

    inlyr.commitChanges()
    iface.vectorLayerTools().stopEditing(inlyr)
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    inlyr = None

    def calcField_bySelection_double(input_shp, select_fields, select_operators, select_values, condicional, target_field,
target_value):

        inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')

        select = ('"{field1}" {operator1} {value1} {cond} "{field2}" {operator2} {value2}').format(
            field1 = select_fields[0],
            operator1 = select_operators[0],
            value1 = select_values[0],
            cond = condicional,
            field2 = select_fields[1],
            operator2 = select_operators[1],
            value2 = select_values[1]
        )
        inlyr.selectByExpression(select)

        ids = inlyr.selectedFeatureIds()
        request = QgsFeatureRequest()
        request.setFilterFids(ids)

        for feature in inlyr.getFeatures(request):
            inlyr.startEditing()
            feature.setAttribute(feature.fieldNameIndex(target_field), target_value)
            inlyr.updateFeature(feature)

        inlyr.commitChanges()
        iface.vectorLayerTools().stopEditing(inlyr)
        inlyr = None

    def calcField_bySelection_triple(input_shp, select_fields, select_operators, select_values, condicional, target_field,
target_value):

        inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')

        select = ('"{field1}" {operator1} {value1} {cond1} "{field2}" {operator2} {value2} {cond2} "{field3}" {operator3}
{value3}').format(
            field1 = select_fields[0],
            operator1 = select_operators[0],
            value1 = select_values[0],
            cond1 = condicional[0],
            field2 = select_fields[1],
            operator2 = select_operators[1],
            value2 = select_values[1],
            cond2 = condicional[1],
            field3 = select_fields[2],
            operator3 = select_operators[2],
            value3 = select_values[2]
        )
        inlyr.selectByExpression(select)

        ids = inlyr.selectedFeatureIds()
        request = QgsFeatureRequest()
        request.setFilterFids(ids)

        for feature in inlyr.getFeatures(request):
            inlyr.startEditing()
            feature.setAttribute(feature.fieldNameIndex(target_field), target_value)
            inlyr.updateFeature(feature)

        inlyr.commitChanges()
        iface.vectorLayerTools().stopEditing(inlyr)
        inlyr = None

    def calcField_Clear_doubleSub(input_shp, calc_fields, target_field):

        inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')

        for feature in inlyr.getFeatures():
            inlyr.startEditing()
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            feature.setAttribute(feature.fieldNameIndex(target_field), (feature[calc_fields[1]]-feature[calc_fields[0]]))
            inlyr.updateFeature(feature)

        inlyr.commitChanges()
        iface.vectorLayerTools().stopEditing(inlyr)
        inlyr = None

    def symmetrical_difference(input_shp, overlay, output_shp):
        inlyr = QgsVectorLayer(input_shp, 'temp', 'ogr')
        processing.run("qgis:symmetricaldifference", {"INPUT": input_shp, "OVERLAY": overlay, "OUTPUT": output_shp})
        del inlyr
        return(output_shp)

    def joinLocation(in_layer, join_layer, join_field, output_shp):
        inlyr = QgsVectorLayer(in_layer, 'temp', 'ogr')
        inlyrJ = QgsVectorLayer(join_layer, 'temp', 'ogr')
        #context = dataobjects.createContext()
        #context.setInvalidGeometryCheck(QgsFeatureRequest.GeometryNoCheck)
        processing.run("native:joinattributesbylocation", {"INPUT": in_layer, "JOIN": join_layer, "PREDICATE": 1, "JOIN_FIELDS":
join_field, "METHOD": 1,"OUTPUT": output_shp})
        del inlyr, inlyrJ
        return(output_shp)

    def selectPref(in_layer, intersect, buff, output):
        inlyr = QgsVectorLayer(in_layer, 'temp', 'ogr')
        inlyrI = QgsVectorLayer(intersect, 'temp', 'ogr')
        processing.run('native:buffer', {"INPUT":inlyr, "DISTANCE": 0.2, "OUTPUT": buff})
        del inlyr
        inlyr = QgsVectorLayer(buff, 'temp', 'ogr')
        #processing.run('qgis:selectbylocation', {"INPUT":inlyr, "PREDICATE": 3, "INTERSECT": inlyrI, "METHOD": 0})
        #processing.run('qgis:selectbylocation', {"INPUT":inlyr, "PREDICATE": 4, "INTERSECT": inlyrI, "METHOD": 1})
        #inlyr.invertSelection()
        processing.run('qgis:selectbylocation', {"INPUT":inlyr, "PREDICATE": 6, "INTERSECT": inlyrI, "METHOD": 0})
        processing.run('native:saveselectedfeatures', {"INPUT":inlyr, "OUTPUT": output})
        #driver = ogr.GetDriverByName('ESRI Shapefile')
        #outds = driver.CreateDataSource(output)
        #outlyr = outds.CopyLayer(inlyr, output)
        del inlyr, inlyrI, buff
        #return(outlyr)




    feedback.pushInfo("This process may take a long time, please wait.")
    feedback.pushInfo("If a command window opens, do not close it. It will close automatically.")

    if not self.dstr:
        # Fishnet Area and CRS
        fishnet_t1 = create_fishnet(study_areaS, fishnet_temp1, squares, squares)
        feedback.pushInfo("Fishnet Created")

        my_crs = QgsCoordinateReferenceSystem(fishnet_t1[1])
        QgsProject.instance().setCrs(my_crs)
        QgsProject.instance().setEllipsoid("NONE")

        feedback.pushInfo("Clipping Fishnet...")
        study_areaS = clip_shp(fishnet_t1[0], study_areaS, fishnet_temp2)
        feedback.pushInfo("Fishnet Clipped by Study Area")

        # Intersect moment 1, 2 and 3
        intersect_t1 = intersect_simple(land1S, study_areaS, intersect_temp1)
        update_field_area(intersect_t1, areaInicial_field)
        feedback.pushInfo("Intersect 1 Done")

        intersect_t2 = intersect_simple(land2S, study_areaS, intersect_temp2)
        update_field_area(intersect_t2, areaInicial_field)
        feedback.pushInfo("Intersect 2 Done")
        if self.m3:
            intersect_t3 = intersect_simple(land3S, study_areaS, intersect_temp3)
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        update_field_area(intersect_t3, areaInicial_field)
        feedback.pushInfo("Intersect 3 Done")
else:
    # Create FID field
    create_field(study_areaS, Fid_Field, 0, 0, 10)
    update_field_fid(study_areaS, Fid_Field)

    # Districts CRS
    layer = QgsVectorLayer(study_areaS, 'temp', 'ogr')
    lyrCRS = layer.crs().authid()
    my_crs = QgsCoordinateReferenceSystem(lyrCRS)
    QgsProject.instance().setCrs(my_crs)
    QgsProject.instance().setEllipsoid("NONE")

    # Intersect moment 1, 2 and 3
    intersect_t1 = intersect_simple(land1S, study_areaS, intersect_temp1)
    update_field_area(intersect_t1, areaInicial_field)
    feedback.pushInfo("Intersect 1 Done")

    intersect_t2 = intersect_simple(land2S, study_areaS, intersect_temp2)
    update_field_area(intersect_t2, areaInicial_field)
    feedback.pushInfo("Intersect 2 Done")

    if self.m3:
        intersect_t3 = intersect_simple(land3S, study_areaS, intersect_temp3)
        update_field_area(intersect_t3, areaInicial_field)
        feedback.pushInfo("Intersect 3 Done")

# Multipart to singlepart moment 1, 2 and 3
multisingle_t1 = multi_to_single(intersect_t1, multisingle_temp1)
update_field_area(multisingle_t1, areaInicial_field)
feedback.pushInfo("Multi to SinglePart 1 Done")

multisingle_t2 = multi_to_single(intersect_t2, multisingle_temp2)
update_field_area(multisingle_t2, areaInicial_field)
feedback.pushInfo("Multi to SinglePart 2 Done")

if self.m3:
    multisingle_t3 = multi_to_single(intersect_t3, multisingle_temp3)
    update_field_area(multisingle_t3, areaInicial_field)
    feedback.pushInfo("Multi to SinglePart 3 Done")

# Feature selection moment 1, 2 and 3
select_t1 = select_att(multisingle_t1, areaInicial_field, ">=", min_size, select_temp1)
update_field_area(select_t1, areaInicial_field)
feedback.pushInfo("Patch 1 Done")

select_t2 = select_att(multisingle_t2, areaInicial_field, ">=", min_size, select_temp2)
update_field_area(select_t2, areaInicial_field)
feedback.pushInfo("Patch 2 Done")

if self.m3:
    select_t3 = select_att(multisingle_t3, areaInicial_field, ">=", min_size, select_temp3)
    update_field_area(select_t3, areaInicial_field)
    feedback.pushInfo("Patch 3 Done")

# Join fields moment 1, 2 and 3
statsSum_1 = stats_by_AreaFID(select_t1, areaInicial_field, Fid_Field, area_1, freq_field_1)
create_field(study_areaS, freq_field_1, 0, 0, 32)
create_field(study_areaS, area_1, 1, 6, 32)
join_FID_AREA(study_areaS, statsSum_1, Fid_Field, freq_field_1, area_1)
feedback.pushInfo("Join fields 1 Done")

statsSum_2 = stats_by_AreaFID(select_t2, areaInicial_field, Fid_Field, area_2, freq_field_2)
create_field(study_areaS, freq_field_2, 0, 0, 32)
create_field(study_areaS, area_2, 1, 6, 32)
join_FID_AREA(study_areaS, statsSum_2, Fid_Field, freq_field_2, area_2)
feedback.pushInfo("Join fields 2 Done")

if self.m3:
    statsSum_3 = stats_by_AreaFID(select_t3, areaInicial_field, Fid_Field, area_3, freq_field_3)
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    create_field(study_areaS, freq_field_3, 0, 0, 32)
    create_field(study_areaS, area_3, 1, 6, 32)
    join_FID_AREA(study_areaS, statsSum_3, Fid_Field, freq_field_3, area_3)
    feedback.pushInfo("Join fields 3 Done")

# Symmetrical Difference moment 1, 2 and 3
symmetrical_difference(select_t1, select_t2, symdif12)
feedback.pushInfo("Symmetrical Difference 12 Done")
multisingle_symdif12 = multi_to_single(symdif12, ms_symdif12)
feedback.pushInfo("Multi to SinglePart 12 Done")
create_field(multisingle_symdif12, sym12, 0, 0, 1)
calcField(multisingle_symdif12, sym12, 1, Fid_Field, "FID_2")
#processing.run('native:fixgeometries', {"INPUT":multisingle_symdif12, "OUTPUT": symdif12_fix})

if pref:
    feedback.pushInfo("Perforation 12 check")
    selectPref(multisingle_symdif12, select_t1, prefBuff12, pref12)

fishnet_t3 = joinLocation(study_areaS, multisingle_symdif12, sym12, fishnet_temp3)

if self.m3:
    symmetrical_difference(select_t2, select_t3, symdif23)
    feedback.pushInfo("Symmetrical Difference 23 Done")
    multisingle_symdif23 = multi_to_single(symdif23, ms_symdif23)
    feedback.pushInfo("Multi to SinglePart 23 Done")
    create_field(multisingle_symdif23, sym23, 0, 0, 1)
    calcField(multisingle_symdif23, sym23, 1, Fid_Field, "FID_2")

    if pref:
        feedback.pushInfo("Perforation 23 check")
        selectPref(multisingle_symdif23, select_t2, prefBuff23, pref23)

    fishnet_t4 = joinLocation(fishnet_t3, multisingle_symdif23, sym23, fishnet_temp4)

    symmetrical_difference(select_t1, select_t3, symdif13)
    feedback.pushInfo("Symmetrical Difference 13 Done")
    multisingle_symdif13 = multi_to_single(symdif13, ms_symdif13)
    feedback.pushInfo("Multi to SinglePart 13 Done")
    create_field(multisingle_symdif13, sym13, 0, 0, 1)
    calcField(multisingle_symdif13, sym13, 1, Fid_Field, "FID_2")

    if pref:
        feedback.pushInfo("Perforation 13 check")
        selectPref(multisingle_symdif13, select_t1, prefBuff13, pref13)

    fishnet_t5 = joinLocation(fishnet_t4, multisingle_symdif13, sym13, fishnet_temp5)

# Fishnet evaluation
feedback.pushInfo("Finishing the process, please wait...")
if not self.m3:
    if pref:
        create_field(fishnet_t3, perforation12, 0, 0, 5)
    create_field(fishnet_t3, presence_1, 0, 0, 5)
    create_field(fishnet_t3, presence_2, 0, 0, 5)
    create_field(fishnet_t3, var_area_12, 1, 6, 32)
    create_field(fishnet_t3, var_NP_12, 0, 0, 32)
    create_field(fishnet_t3, ToD_12, 2, 0, 50)

    if pref:
        calcField_byLocation(fishnet_t3, pref12, perforation12, 1)
    calcField_bySelection_single(fishnet_t3, freq_field_1, ">", 0, presence_1, 1)
    calcField_bySelection_single(fishnet_t3, freq_field_1, "=", 0, presence_1, 0)
    calcField_bySelection_single(fishnet_t3, freq_field_2, ">", 0, presence_2, 1)
    calcField_bySelection_single(fishnet_t3, freq_field_2, "=", 0, presence_2, 0)

    calcField_Clear_doubleSub(fishnet_t3, [area_1, area_2], var_area_12)
    calcField_Clear_doubleSub(fishnet_t3, [freq_field_1, freq_field_2], var_NP_12)
    calcField_bySelection_double(fishnet_t3, [var_area_12, var_area_12], [">=", "<="], [-100, 100], "AND", var_area_12, 0)

    calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], ["=", "="], [0, 0], "AND", ToD_12, "A - No change")
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        calcField_bySelection_triple(fishnet_t3, [var_area_12, var_NP_12, sym12], ["=", "=", "="], [0, 0, 1], ["AND", "AND"], ToD_12,
"A1 - Spatial shift")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], ["=", ">"], [0, 0], "AND", ToD_12, "B - Fragmentation
per se")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], ["=", "<"], [0, 0], "AND", ToD_12, "C - Aggregation per
se")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], [">", "="], [0, 0], "AND", ToD_12, "D - Gain")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], ["<", "="], [0, 0], "AND", ToD_12, "E - Loss")
        if pref:
          calcField_bySelection_triple(fishnet_t3, [var_area_12, var_NP_12, perforation12], ["<", "=", "="], [0, 0, 1], ["AND",
"AND"], ToD_12, "E1 - Perforation")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], [">", ">"], [0, 0], "AND", ToD_12, "F - NP increment by
gain")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], [">", "<"], [0, 0], "AND", ToD_12, "G - Aggregation by
gain")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], ["<", "<"], [0, 0], "AND", ToD_12, "H - NP decrement by
loss")
        calcField_bySelection_double(fishnet_t3, [var_area_12, var_NP_12], ["<", ">"], [0, 0], "AND", ToD_12, "I - Fragmentation by
loss")
        calcField_bySelection_double(fishnet_t3, [presence_1, presence_2], ["=", "="], [0, 0], "AND", ToD_12, "Study object is
absent")

      layer = QgsVectorLayer(fishnet_temp3, "", "ogr")
    else:
      if pref:
        create_field(fishnet_t5, perforation12, 0, 0, 5)
        create_field(fishnet_t5, perforation23, 0, 0, 5)
        create_field(fishnet_t5, perforation13, 0, 0, 5)
      create_field(fishnet_t5, presence_1, 0, 0, 5)
      create_field(fishnet_t5, presence_2, 0, 0, 5)
      create_field(fishnet_t5, presence_3, 0, 0, 5)
      create_field(fishnet_t5, var_area_12, 1, 6, 32)
      create_field(fishnet_t5, var_area_23, 1, 6, 32)
      create_field(fishnet_t5, var_area_13, 1, 6, 32)
      create_field(fishnet_t5, var_NP_12, 0, 0, 32)
      create_field(fishnet_t5, var_NP_23, 0, 0, 32)
      create_field(fishnet_t5, var_NP_13, 0, 0, 32)
      create_field(fishnet_t5, ToD_12, 2, 0, 50)
      create_field(fishnet_t5, ToD_23, 2, 0, 50)
      create_field(fishnet_t5, ToD_13, 2, 0, 50)

      if pref:
        calcField_byLocation(fishnet_t5, pref12, perforation12, 1)
        calcField_byLocation(fishnet_t5, pref23, perforation23, 1)
        calcField_byLocation(fishnet_t5, pref13, perforation13, 1)
      calcField_bySelection_single(fishnet_t5, freq_field_1, ">", 0, presence_1, 1)
      calcField_bySelection_single(fishnet_t5, freq_field_1, "=", 0, presence_1, 0)
      calcField_bySelection_single(fishnet_t5, freq_field_2, ">", 0, presence_2, 1)
      calcField_bySelection_single(fishnet_t5, freq_field_2, "=", 0, presence_2, 0)
      calcField_bySelection_single(fishnet_t5, freq_field_3, ">", 0, presence_3, 1)
      calcField_bySelection_single(fishnet_t5, freq_field_3, "=", 0, presence_3, 0)

      calcField_Clear_doubleSub(fishnet_t5, [area_1, area_2], var_area_12)
      calcField_Clear_doubleSub(fishnet_t5, [area_2, area_3], var_area_23)
      calcField_Clear_doubleSub(fishnet_t5, [area_1, area_3], var_area_13)
      calcField_Clear_doubleSub(fishnet_t5, [freq_field_1, freq_field_2], var_NP_12)
      calcField_Clear_doubleSub(fishnet_t5, [freq_field_2, freq_field_3], var_NP_23)
      calcField_Clear_doubleSub(fishnet_t5, [freq_field_1, freq_field_3], var_NP_13)
      calcField_bySelection_double(fishnet_t5, [var_area_12, var_area_12], [">=", "<="], [-100, 100], "AND", var_area_12, 0)
      calcField_bySelection_double(fishnet_t5, [var_area_23, var_area_23], [">=", "<="], [-100, 100], "AND", var_area_23, 0)
      calcField_bySelection_double(fishnet_t5, [var_area_13, var_area_13], [">=", "<="], [-100, 100], "AND", var_area_13, 0)

      calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], ["=", "="], [0, 0], "AND", ToD_12, "A - No change")
      calcField_bySelection_triple(fishnet_t5, [var_area_12, var_NP_12, sym12], ["=", "=", "="], [0, 0, 1], ["AND", "AND"], ToD_12,
"A1 - Spatial shift")
      calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], ["=", ">"], [0, 0], "AND", ToD_12, "B - Fragmentation
per se")
      calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], ["=", "<"], [0, 0], "AND", ToD_12, "C - Aggregation per
se")
      calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], [">", "="], [0, 0], "AND", ToD_12, "D - Gain")
      calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], ["<", "="], [0, 0], "AND", ToD_12, "E - Loss")
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        if pref:
            calcField_bySelection_triple(fishnet_t5, [var_area_12, var_NP_12, perforation12], ["<", "=", "="], [0, 0, 1], ["AND",
"AND"], ToD_12, "E1 - Perforation")
        calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], [">", ">"], [0, 0], "AND", ToD_12, "F - NP increment by
gain")
        calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], [">", "<"], [0, 0], "AND", ToD_12, "G - Aggregation by
gain")
        calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], ["<", "<"], [0, 0], "AND", ToD_12, "H - NP decrement by
loss")
        calcField_bySelection_double(fishnet_t5, [var_area_12, var_NP_12], ["<", ">"], [0, 0], "AND", ToD_12, "I - Fragmentation by
loss")
        calcField_bySelection_double(fishnet_t5, [presence_1, presence_2], ["=", "="], [0, 0], "AND", ToD_12, "Study object is
absent")

        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], ["=", "="], [0, 0], "AND", ToD_23, "A - No change")
        calcField_bySelection_triple(fishnet_t5, [var_area_23, var_NP_23, sym23], ["=", "=", "="], [0, 0, 1], ["AND", "AND"], ToD_23,
"A1 - Spatial shift")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], ["=", ">"], [0, 0], "AND", ToD_23, "B - Fragmentation
per se")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], ["=", "<"], [0, 0], "AND", ToD_23, "C - Aggregation per
se")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], [">", "="], [0, 0], "AND", ToD_23, "D - Gain")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], ["<", "="], [0, 0], "AND", ToD_23, "E - Loss")
        if pref:
            calcField_bySelection_triple(fishnet_t5, [var_area_23, var_NP_23, perforation23], ["<", "=", "="], [0, 0, 1], ["AND",
"AND"], ToD_23, "E1 - Perforation")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], [">", ">"], [0, 0], "AND", ToD_23, "F - NP increment by
gain")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], [">", "<"], [0, 0], "AND", ToD_23, "G - Aggregation by
gain")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], ["<", "<"], [0, 0], "AND", ToD_23, "H - NP decrement by
loss")
        calcField_bySelection_double(fishnet_t5, [var_area_23, var_NP_23], ["<", ">"], [0, 0], "AND", ToD_23, "I - Fragmentation by
loss")
        calcField_bySelection_double(fishnet_t5, [presence_2, presence_3], ["=", "="], [0, 0], "AND", ToD_23, "Study object is
absent")

        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], ["=", "="], [0, 0], "AND", ToD_13, "A - No change")
        calcField_bySelection_triple(fishnet_t5, [var_area_13, var_NP_13, sym13], ["=", "=", "="], [0, 0, 1], ["AND", "AND"], ToD_13,
"A1 - Spatial shift")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], ["=", ">"], [0, 0], "AND", ToD_13, "B - Fragmentation
per se")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], ["=", "<"], [0, 0], "AND", ToD_13, "C - Aggregation per
se")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], [">", "="], [0, 0], "AND", ToD_13, "D - Gain")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], ["<", "="], [0, 0], "AND", ToD_13, "E - Loss")
        if pref:
            calcField_bySelection_triple(fishnet_t5, [var_area_13, var_NP_13, perforation13], ["<", "=", "="], [0, 0, 1], ["AND",
"AND"], ToD_13, "E1 - Perforation")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], [">", ">"], [0, 0], "AND", ToD_13, "F - NP increment by
gain")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], [">", "<"], [0, 0], "AND", ToD_13, "G - Aggregation by
gain")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], ["<", "<"], [0, 0], "AND", ToD_13, "H - NP decrement by
loss")
        calcField_bySelection_double(fishnet_t5, [var_area_13, var_NP_13], ["<", ">"], [0, 0], "AND", ToD_13, "I - Fragmentation by
loss")
        calcField_bySelection_double(fishnet_t5, [presence_1, presence_3], ["=", "="], [0, 0], "AND", ToD_13, "Study object is
absent")

        layer = QgsVectorLayer(fishnet_temp5, "", "ogr")

    # Finish process
    (sink, dest_id) = self.parameterAsSink(
        parameters,
        self.OUTPUT,
        context,
        layer.fields(),
        layer.wkbType(),
        layer.sourceCrs()
    )
```

```python
    for f in layer.getFeatures():
        sink.addFeature(f, QgsFeatureSink.FastInsert)

    QgsProject.instance().removeMapLayer(layer.id())
    if not self.dstr:
        del fishnet_t1
    del layer, study_areaS, fishnet_t3
    if self.m3:
        del fishnet_t4, fishnet_t5


    for root, dirs, files in os.walk(dirTemp):
        for file in files:
            path = os.path.join(dirTemp, file)
            os.remove(path)
    os.rmdir(dirTemp)

    self.dest_id=dest_id

    return {self.OUTPUT: dest_id}

def postProcessAlgorithm(self, context, feedback):
    output = QgsProcessingUtils.mapLayerFromString(self.dest_id, context)
    path=ProcessingConfig.getSetting('SCRIPTS_FOLDERS') + '\LDT4QGIS_2M_symbology.qml'
    if self.m3:
        path=ProcessingConfig.getSetting('SCRIPTS_FOLDERS') + '\LDT4QGIS_3M_symbology.qml'
    output.loadNamedStyle(path)
    output.triggerRepaint()
    return {self.OUTPUT: self.dest_id}
```

APPENDIX IV – Published articles

Assessment of landscape composition and configuration via spatial metrics combination: conceptual framework proposal and method improvement

Rui Machado, Sérgio Godinho, Janez Pirnat, Nuno Neves & Pedro Santos

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Assessment of landscape composition and configuration via spatial metrics combination: conceptual framework proposal and method improvement

Rui Machado[a] , Sérgio Godinho[a], Janez Pirnat[b] , Nuno Neves[c,d] and Pedro Santos[e]

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ABSTRACT

Landscape transformations have been a central topic in landscape ecology due to their influence on ecosystem services and biodiversity conservation. Landscape dynamics can be very complex and one of the most difficult issues to analyse is the contribution and the effects of landscape composition and configuration changes to the overall impact. We enhanced an existing approach designed to assess landscape transformation via metrics combination. Based on the area and the number of patches, we defined a Landscape Dynamics Typology (LDT) with eight different dynamics. The agro-forestry areas dynamics between 1990 and 2006, in Portugal, is presented as a practical example to show the implementation steps and the outputs obtained. A diagnostic tree is provided to help identify which dynamics are present. This new method provides detailed results concerning the location of the different dynamics within a landscape. The LDT is a versatile tool and is able to accommodate upgrades.

Introduction

Human-induced landscape changes often have important drawbacks such as habitat loss, degradation and fragmentation that have been implicated as the major causes of the recent biodiversity loss worldwide (Jaeger, Madrinan, Soukup, Schwick, & Kienast, 2011; Millennium Ecosystem Assessment, 2005). For that reason, habitat fragmentation has been widely regarded as a central issue in conservation biology (Saunders, Hobbs, & Margules, 1991; Wiens, 1996; Wilcove, McLellan, & Dobson, 1986). Natural landscapes can support high levels of biodiversity and also provide vital ecosystem services. Habitat loss and fragmentation are also responsible for diminishing ecosystems' capacity to provide such services (Millennium Ecosystem Assessment, 2005).

Fragmentation per se has been defined as 'the breaking up of a habitat, ecosystem or land-use type into smaller parcels' (Forman, 1995). Note that *fragmentation* is the process and the object can be habitat (which is species-specific), land cover or land use types, landscapes, ecosystems. In the same

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line of thought, habitat loss is the difference between two amounts of habitats for a given species (more at first and less at the end). Not only do habitat fragmentation and habitat loss co-occur in a landscape, they also interact and depend on each other, making it difficult to isolate their independent effects (Fischer & Lindenmayer, 2007). Nevertheless, this separation is necessary to better understand their impacts on the landscape. Results from empirical studies on habitat fragmentation are frequently difficult to interpret and compare because measurements are made such that habitat loss and habitat fragmentation are not separated (Fahrig, 2003). Habitat loss, as noted by Fahrig (1997), is the main process driving landscape transformation, and yet it has received less research attention than expected (Fazey, Fischer, & Lindenmayer, 2005) especially since it is considered by many to be the leading cause of decline in native species (Fahrig, 1997; Foley et al., 2005).

Many indices have been developed to assess spatial arrangement of habitat patches, classes of patches and landscape characteristics like dominance, diversity, contagion and fractal dimension (Haines-Young & Chopping, 1996; Turner, 1989), but few were specifically suited to measure fragmentation. An exception to that are the three metrics introduced by Jaeger (2000) that characterise landscape fragmentation in a geometric perspective: degree of landscape division, splitting index and effective mesh size. Long, Nelson, and Wulder (2010) highlighted the incorrect approaches that measure forest fragmentation considering forest loss and configurational change together because their ecological responses are different. Consequently, they developed a measure (proportion of landscape displacement from configuration) to quantify the relative contributions of forest loss and configurational change to the so-called forest fragmentation. Bogaert, Ceulemans, and Salvador-Van Eysenrode (2004) defined 10 processes responsible for pattern change, looking not only at aspects related to habitat loss but also included transformation processes that cause expansion of the land cover of interest (habitat gain). The need to correctly assess the landscape dynamics distinguishing the effects of habitat fragmentation from those of habitat loss have been considered in recent studies (e.g. Johnstone, Lill, & Reina, 2014; Plećaš et al., 2014; Sauder & Rachlow, 2014; Steckel et al., 2014). This paper was inspired by the work of Bogaert et al. (2004) as the goals are similar and the methods follow those presented in their paper. We developed their foundations by introducing some variations and expanding the application possibilities. The objective of this paper was to propose a method for assessing habitat composition and configuration changes separating the processes of habitat loss/gain and habitat fragmentation/aggregation.

Our main contributions are:

- The establishment of a 'Landscape Dynamics Typology' (LDT). This is a list of possible types of *dynamics* (transformation patterns) that can take place in a landscape ('Landscape Dynamics Typology' subtopic in the 'Results' section);
- The supply of a diagnostic tree, which is a valuable tool to easily find out which dynamics are taking place in a focal area;
- The presentation of a case study in Portugal, illustrating how the LDT can be applied in landscape analysis. Even though the method is designed to deliver an overall result (landscape-level analysis), which represents a broad simplification of the processes involved, we show how to use analytical units to obtain located and more detailed results ('Case study' topic in the 'Results' section).

Materials and methods

Conceptual framework and terminology clarification

A new conceptual framework is needed because some terms and concepts in ecology have become vague due to inaccurate use. Designations like 'habitat fragmentation' have become uncritically used (Lindenmayer & Fischer, 2007) and 'habitat' has become a vague concept (Mitchell & Powell, 2003).

In order to avoid misapplication of concepts and to minimise semantic confusion, in this section we define the study's object and clarify some terms adopted. 'Fragmentation' has its opposite which is 'aggregation' and 'loss' has the opposite aspect 'gain'. As to the term 'habitat', it has been used in very different contexts, with different meanings, not always keeping in mind that 'habitat' is a species-specific

concept and therefore should be used in species-specific based studies (St-Laurent, Dussault, Ferron, & Gagnon, 2009) and should not be equated with native vegetation (Fischer & Lindenmayer, 2007).

We use the designations 'Geometric Variation', which comprises fragmentation and aggregation, and 'Amount Variation' which includes loss and gain. The former represents the configuration changes and the latter represents the composition changes in the landscape. We do not use the term 'habitat'. The correct term to use depends on the study context. It can be 'habitat', 'landscape', 'biotope', 'land cover class' or others, but it is important to be precise. In the case study presented in this paper, we worked with agro-forestry areas.

Landscape metric selection

Landscape metrics measure and in various ways describe the spatial structure of patches, patch classes or landscapes (Leitão, Miller, Ahern, & McGarigal, 2006). This capacity is relevant for ecological studies and thus plenty of metrics have emerged in the last two decades making it a complex choice regarding which ones are the best for specific analysis (Cushman, McGarigal, & Neel, 2008). Cushman et al. (2008) stated that landscape structure must be quantified using the smallest number of independent metrics possible. In this work, we use the land cover class amount (area in hectares) to measure composition and the number of patches (NP) to assess configuration. Class area is a measure of landscape composition which reveals how much of the landscape is comprised by a particular patch type and NP which represents a measure of the subdivision of the patch type and is linked with a number of ecological processes (Cumming & Vernier, 2002; McGarigal, Cushman, & Ene, 2012).

LDT establishment

LDT is a classification system in which landscape changes are aggregated according to the processes that give rise to the changes. The different dynamics are defined for a binary landscape (with two classes only: the land cover under study and other class that encompasses the different land covers) and obtained by considering how eventual composition and configuration changes are reflected by the two selected metrics, area and NP. Some dynamics are relatively easy to understand:

(i) If, between two instants, the amount remains the same and the NP increases we are facing a fragmentation per se event (pure geometric variation).

(ii) If, between two instants, the amount increases and NP decreases, it means that we are in the presence of an aggregation of patches due to an amount gain (geometric variation caused by an amount variation).

However, there are other patterns that reflect different metric combinations that are conceptually difficult to interpret. To answer elementary questions that arise after the analysis, such as 'How far can this trend go?' or 'How will the landscape look if the agro-forestry areas increase?' it is necessary to develop a moderately sophisticated forecast tool or protocol. We take the first step for this path through an approach based on the premise that the trend will continue. Using sampling units such as squares, we assume, for convenience, that the squares with amount gain will continue gaining, leading to a 'total cover' scenario, and the squares with amount loss will keep losing, eventually reaching a 'no cover' situation. In each of these two paths towards the extremes, the squares can pass through different dynamics.

Finally, a dichotomous tree key was also created in order to assist the user to classify a dynamic in a prompt and easy way.

Case study, land cover data and procedures

To illustrate our approach, we studied the spatiotemporal dynamics of the agro-forestry areas in the Portuguese continental territory between the years 1990 and 2006. Agro-forestry areas were obtained
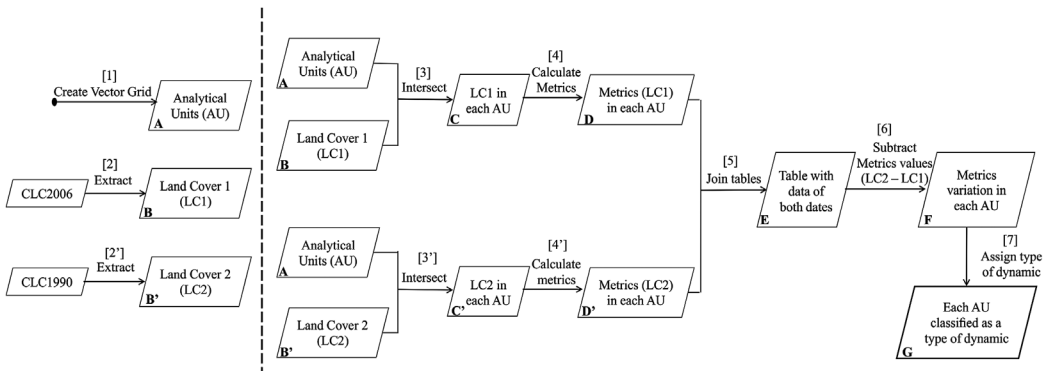
Figure 1. LDT implementation steps.

using ArcGIS 10.2 ™ (ESRI, USA) software and three shapefiles available online: two CORINE land cover shapefiles (years 1990 and 2006) (Caetano, Nunes, & Nunes, 2009) and the Portuguese Administrative Boundaries Official Map—2014 version. According to the CORINE land cover nomenclature, the agro-forestry areas encompass annual crops or grazing land under the wooded cover of forestry species. This makes this land cover category not completely equivalent, but greatly coincident to one of the most important ecosystems in the Iberian Peninsula, the *montado*, in Portugal and *dehesa*, in Spain. The *montado* has suffered a decrease in the last decades (Godinho et al., 2016), which make it suitable to demonstrate our LDT method.

The ultimate task was to compare values of the metrics in the two moments (1990 and 2006) in order to be able to assign a dynamic. Aiming to do this not only globally (at the landscape scale) but also at a more detailed scale, a sampling scheme was needed. For that, we built a 10×10 km grid and used each square as an analytical unit, where area and NP of agro-forestry land cover category were computed. In summary, this task involved seven steps: (i) building a 10×10 km grid; (ii) extracting the agro-forestry areas from both CORINE land cover shapefiles; (iii) representing the presence of agro-forestry areas (for both years) in each square; (iv) calculating the metrics (Area and NP) of the agro-forestry areas (for both years) in each square; (v) joining the results of both calculations into a single table; (vi) calculating the variation between years for each metric and in each square; (vii) assigning each square to a dynamic (Figure 1). Selection and calculation tools (e.g. 'select by attributes' and 'field calculator') were used to automate the process avoiding the need to replicate the steps for each square.

Results

There are two major categories of results: those concerning the LDT and those regarding the practical implementation.

Landscape Dynamics Typology

The LDT is a list of possible landscape changes, defined by the metrics' behaviour. Altogether it contains the dynamics, its designations, the metrics' behaviour and a graphic representation of the change. The dynamics were integrated into a dichotomous tree, useful for a quick analysis of the transformation processes acting in the landscape (Figure 2). The tree has two main levels of analysis. In the first, we must check whether the land cover class amount remained the same ($\Delta A = 0$), increased ($\Delta A > 0$) or decreased ($\Delta A < 0$). Having done that, the second task is to verify whether the NP remained the same ($\Delta Np = 0$), increased ($\Delta Np > 0$) or decreased ($\Delta Np < 0$). These two steps will guide us through the tree and lead us to a specific dynamic.
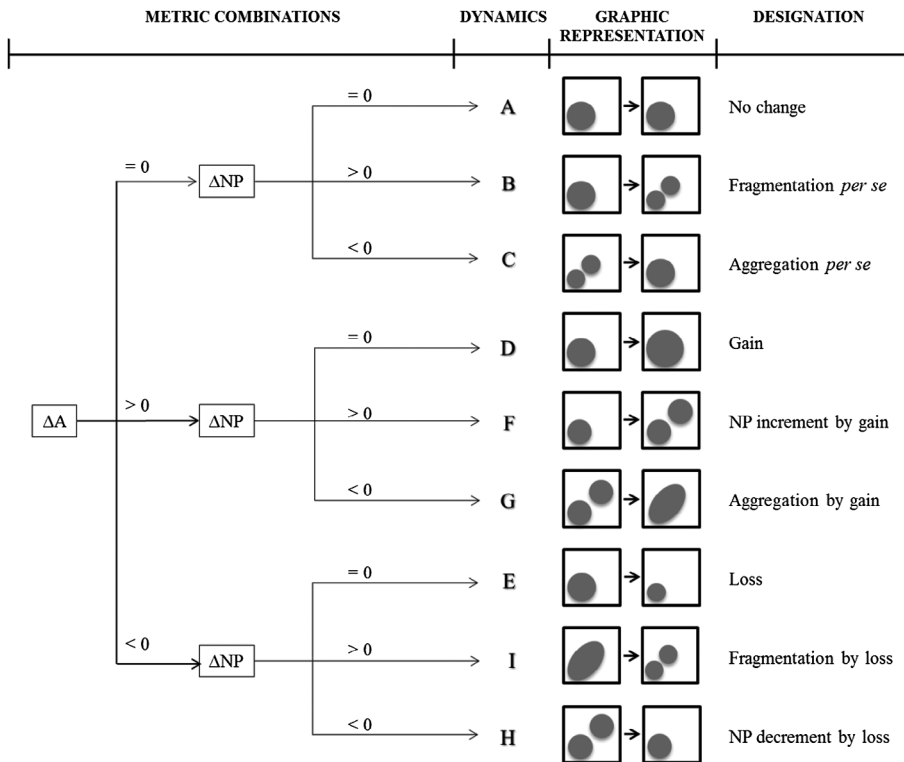
Figure 2. LDT auxiliary diagnostic tree. Dynamics: (A) If there is no change in an area or in the number of patches (NP), we assume that landscape (or the analytical unit extent) did not change; (B) If the area remained the same but the NP increased, it means a fragmentation occurred; (C) If the area remained the same but NP decreased, then an aggregation took place; (D) If the area increased and the NP is equal, it represents a gain of area; (E) If the area decreased and the NP did not change, there is a loss of area; (F) If both area and NP increased, it led to new patch creation; (G) If the area increased and NP decreased, an aggregation occurred due to area gain; (H) If both area and NP decreased, a patch decrement occurred due to area loss; (I) If area decreased and the NP increased, it means that fragmentation occurred due to area loss.

The LDT forecast scheme shows how the dynamics are interconnected in a trajectory for amount gain or loss (Figure 3). In the presence of area gain: dynamic G (aggregation by gain) will continue eventually until it reaches a 'total cover' situation; dynamic F (NP increment by gain) can progress to a point when the amount is such that it will trigger the dynamic G (aggregation by gain); dynamic D (gain) has two possible trajectories before it reaches total cover: if there is only one patch, it can continue growing to the maximum extent; if there are multiple patches, their expansion will at some point create aggregation (dynamic G) and after evolve to total cover.

In the presence of area loss: dynamic H (NP decrement by loss) is going directly to a 'no cover' scenario when the last patch disappears; dynamic I (fragmentation by loss) is a predecessor of the dynamic H, meaning that the fragmentation produces small patches that will disappear thereafter; dynamic E (loss) can go directly towards a 'no cover' situation if there is a single patch in the area, or head to the dynamic H before that, if there are multiple patches. Finally, aggregation per se (dynamic C) and fragmentation per se (dynamic B) are not mentioned in the LDT forecast scheme because both processes only occur when ΔA = 0 (Figure 2).

Case study

From the total 1012 squares, agro-forestry areas in 1990 or 2006 were found in 417. They were present in 416 squares in 1990 and in 413 squares in 2006. Agro-forestry areas were present in both 1990 and
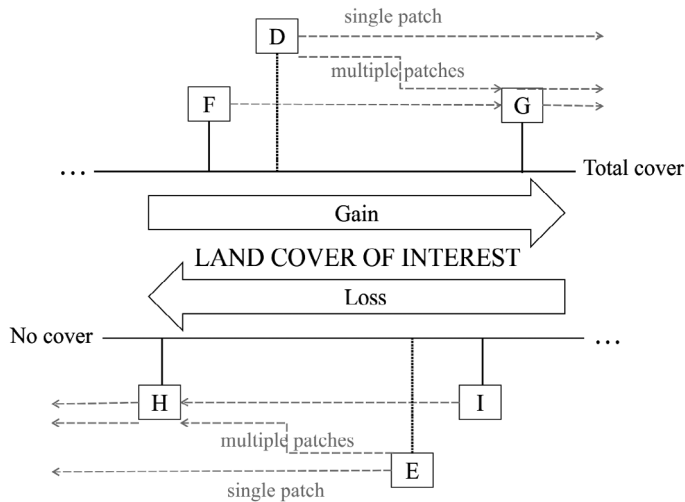
Figure 3. LDT forecast scheme. D—gain; E—loss; F—number of patches increment by gain; G—aggregation by gain; H—number of patches decrement by loss; I—fragmentation by loss.

2006 in 412 squares. In the 1990 map, there were 2977 patches of agro-forestry systems with a total area of 634,862 hectares and in the 2006 map these values were 3028 patches and 621,495 hectares. The global balance shows a variation of +51 patches and −13.358 hectares, suggesting the occurrence of fragmentation due to a loss of agro-forestry amount at the study area scale.

The use of analytical units provides more detailed data such as the number of squares with area loss, gain or maintenance (Figure 4(a)) and the number of squares with NP loss, gain or maintenance (Figure 4(b)). The number of squares displaying negative area variation was much larger than that displaying positive area variation. Almost half of the studied squares registered an agro-forestry areas loss between 1990 and 2006. Regarding the NP variation, the majority of the squares remained unaltered. Besides those, more squares revealed NP increment than decrement, which suggests more fragmentation than aggregation.

Regarding agro-forestry areas variations among squares, an analysis based on area size classes provided new insights about the studied land cover dynamics (Figure 4(c)). The number of squares with loss is much larger than those with gain (200 against 104), which suggest the loss was not confined to a certain region but instead was spread throughout the study area. An additional perspective is provided by Figure 4(d), which shows the number of squares assigned to each dynamic. More than a quarter of the 417 squares with presence of agro-forestry systems showed no changes between the two dates (dynamic A). Pure geometric variations ('aggregation *per se*'—dynamic B and 'fragmentation *per se*'—dynamic C) were not found at all. There were more squares classified as dynamics related to loss (NP decrement by loss—dynamic G, Loss—dynamic H and Fragmentation by loss—dynamic I) than related to gain (NP increment by gain—dynamic D, Gain—dynamic E and Aggregation by gain—dynamic F).

The implementation of the LDT using analytical units took the analysis one step further. It allowed to obtain more detailed results (Figure 4) and to get an insight of where each dynamic was acting in the study area (Figure 5).

Discussion

Landscape Dynamics Typology

Dynamic A symbolises a 'no change' situation that may look improbable in land cover studies but there are a few reasons why a reasonable number of analytical units are assigned to this dynamic: (1) we are
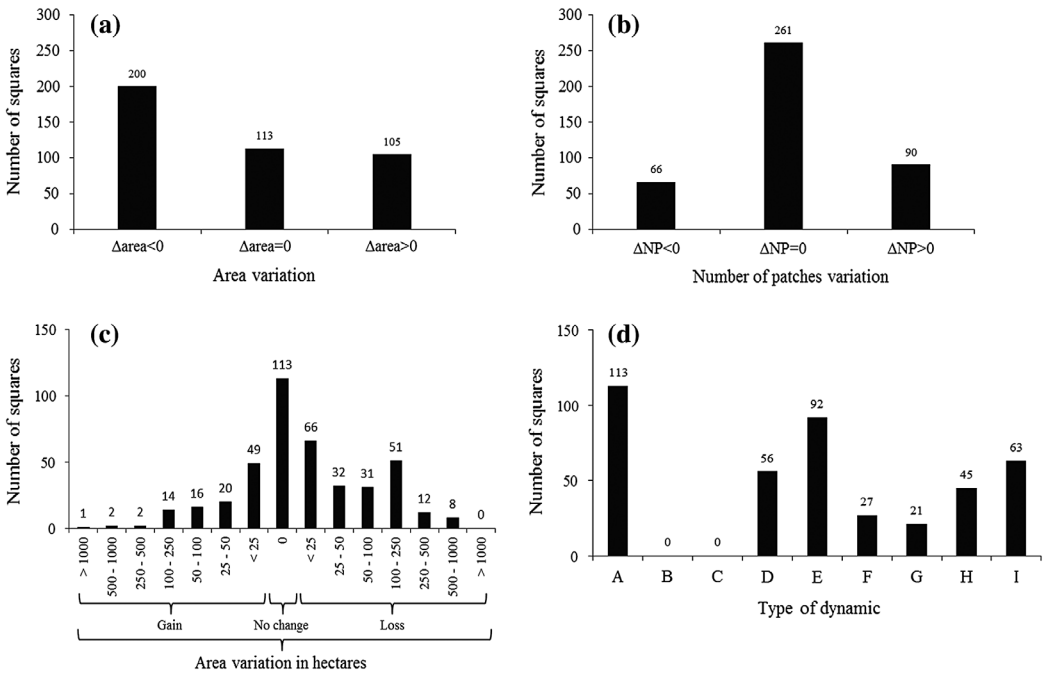
Figure 4. Number of squares vs. area variation (a), area variation with classes (b), number of patches variation (c) and assigned to each dynamic (d); dynamic A 'No change'; dynamic B 'Fragmentation *per se*'; dynamic C 'Aggregation *per se*'; dynamic D 'Gain'; dynamic E 'Loss'; dynamic F 'NP increment by gain'; dynamic G 'Aggregation by gain'; dynamic H 'NP decrement by loss'; dynamic I 'Fragmentation by loss'.

working with binary landscapes, which means only the changes affecting the land cover category of interest are detected; (2) small changes may not be detected due to the minimum map unit or due to generalisation procedures during cartographic production; (3) decisions concerning the minimum patch size to analyse during LDT configuration can also make small changes undetected. In summary, the uncertainty present in the outputs is a function of the cartographic errors (we advise checking the producers reports for error estimation) and the user decisions.

Other than dynamic A, the dynamics can be divided into three classes: (i) geometric or configuration related, (ii) amount or composition related and iii) both geometric and amount related. Dynamics B (fragmentation per se) and C (aggregation per se) represent pure geometric changes. These are not easy to find in nature because they are supposed to occur without any amount change. However, we kept them in the LDT because they have theoretical justification as a landscape could be fragmented without amount change if the patches could be physically rearranged (Wang & Cumming, 2009). Also, it makes sense to test and simulate with these two types of transformations for environmental compensation purposes or ecosystem/habitat mitigation or enhancement. Dynamics D (gain) and E (loss) regard pure amount changes. These are expected to occur frequently and constitute the basis for the dynamics that display both composition and configuration dynamics. Dynamics F (NP increment by gain), G (aggregation by gain), H (NP decrement by loss) and I (fragmentation by loss) reflect geometric changes engendered by amount variations.

The dynamics that we showed in this work are no different from the classical landscape transformations processes recognised in the literature. They are built from the notion of what happens in real landscapes and from theoretical premises. Because we focused markedly on conceptual and terminology clarification, although the underlying processes are the same, our designations may differ from others. The graphical representation for each dynamics (Figure 2) helps to interpret and to compare it to other designations proposed by other authors. We adopted these terms instead of the original for two main
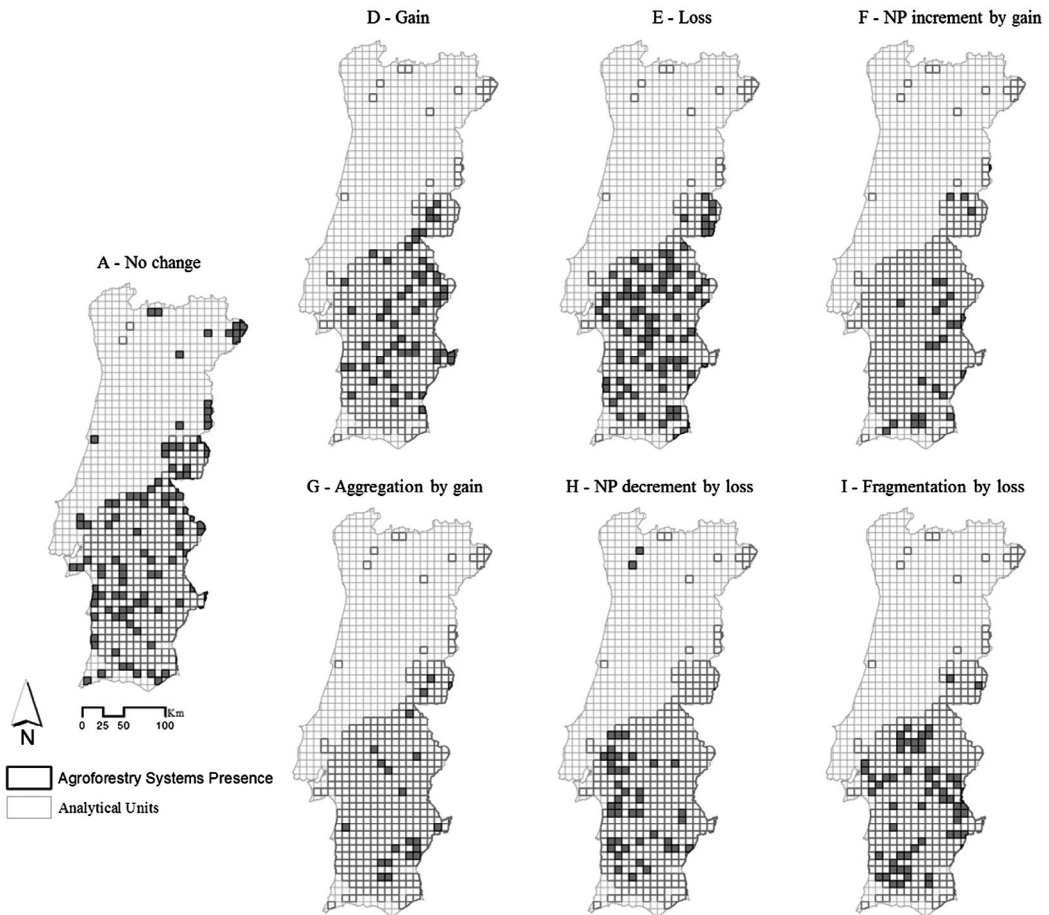
Figure 5. Spatial distribution of different dynamics of landscape dynamics identified between 1990 and 2006.

reasons. The first was to be coherent with the conceptual framework. For instance, we split composition and configuration apart and defined fragmentation per se as a configuration-related process. Knowing that the term 'fragmentation' is usually applied to a transformation which includes amount loss we felt the need to adjust the nomenclature in order to clarify our message. The second reason why we adapted the transformation processes designations was directly linked with the underlying intent of using the LDT in large data-sets via GIS. For that, some simplification/abstraction was needed and some dynamics were not included in the LDT. That was the case for three transformation processes that are usually present in the literature, deformation, shift and perforation, that were absent in the LDT. Deformation is characterised by shape change, shift occurs via patch translocation and perforation is a loss of amount that originates a hole within the patch (Bogaert et al., 2004).

Regarding the scenario forecast possibilities, it is important to stress that LDT does not intend to be a robust and sophisticated scenario simulation tool. One cannot use LDT to thoroughly predict what the metrics values will be on a given date. Yet, it is possible to assume a time sequence to show the trajectory of a landscape or analytical unit. This is possible because it is known that some transformations precede others. For instance, Forman (1995), having defined five processes in land transformation (by order: perforation, dissection, fragmentation, shrinkage and attrition), states that these processes overlap through the period of land transformation beginning with perforation and dissection that evolve to fragmentation, then to shrinkage and finally to attrition. Jaeger (2000) includes a phase called

incision between the perforation and dissection and thus considers six processes. In the same work, the author studies the metrics behaviours by testing them for different fragmentation phases. Because these phases often cannot be separated in real landscapes it is important to work with metrics that behave consistently (increase or decrease continuously) in all phases. Although it may seem the LDT mechanism is different because the detection of changes are reflected on squares being reassigned to a different dynamic, one must keep in mind the dynamics reflect combinations of metrics. For that reason, LDT metrics should be as consistent as possible.

Limitations

Although the dynamics considered here are realistic, such an exercise is necessarily a simplification of reality because landscape dynamics can be very complex and thus hard to fully replicate with this or any other approach. The LDT can provide good outputs and useful information but has limitations and should only be implemented within its range of applicability. One aspect to keep in mind is that this method only interprets binary landscapes and, like every method that does so, it encompasses an abstraction of the landscape complexity, oversimplifying the reality. For instance, the selected metrics would not be able to find 'deformation' or 'shift' because both processes would produce outputs of $\Delta A = 0$ and $\Delta Np = 0$. For that reason, they would be interpreted as 'no change' (dynamic A). Also small changes that occur may not be detected due to mapping errors and end up being assigned to dynamic A. Similarly, 'perforation' wouldn't be distinguishable from the dynamic E ('loss'), as the metrics variations of both would be $\Delta A < 0$ and $\Delta Np = 0$. To track such changes by individually inspecting each square is by no means compatible with the LDT's pragmatic and expedited utilisation perspective. The viable and foreseen alternative is to improve the LDT by integrating new metrics and/or error measures.

LDT is a method to detect changes in the landscape and can be used to study fragmentation at proper scales. Plus, it indicates the composition trend (amount gain or loss) originating from fragmentation or aggregation. Yet, fragmentation, in the broad sense, is a landscape-scale process and should be measured at that scale. For this reason, the use of analytical units in such a study should probably be avoided or at least very well pondered. Paradoxically, one of LDT's strengths, the possibility to apply analytical units, can also bring some weaknesses to the approach. Although there are valuable advantages provided by the use of analytical units, we must highlight the modifiable areal unit problem (MAUP) (Gehlke & Biehl, 1934; Openshaw, 1984). The MAUP refers to the fact that the areal units can be set arbitrarily and are modifiable. Results based on aggregated data following areal units can change if the same data are aggregated under different areal units (Waller & Gotway, 2004). In our example, we used squares as a mask and accounted for the agro-forestry areas in each square. If the squares had a different size and/or a different location, the content in terms of agro-forestry areas would also be different (scale effect and zonation effect, respectively). Long et al., 2010 pointed out the same problem, highlighting that MAUP has impacts in the results, and suggested a multiscale approach when implementing their metric 'proportion of landscape displacement from configuration' (Py) to limit scale-related bias. Simply put, one reality can have numerous representations, even if using the same method, just by varying the analytical unit limits.

Strengths and implementation scope

Looking at the LDT's strengths, we must point out it is simple to understand as it is based on ordinary tools and concepts like landscape metrics and transformation processes well established in the literature.

Concerning applicability we highlight the LDT's versatility in the sense that it systemises a procedure but does not imply the use of specific data formats or software. Besides that, the steps are editable and the parameters adjustable. For instance, the application of analytical units is possible but not required. When using analytical units they can be designed in terms of size, form or location to fit the user's purpose. Finally, it is important to stress how the results are easy to interpret because it all rests

upon metric values and variations. Moreover, using geographic information software it is possible to visualise and to represent the information preserving its spatial attributes.

The LDT was designed to detect landscape transformation processes and thus its range of application is vast. The two most evident applications for LDT are monitoring tasks, in a diagnosis context, or forecasting, in a territory planning framework. Whenever the LDT is well adapted to a certain study subject and context (e.g. a certain land cover class from a specific region, obtained from a given cartography), it can be chosen as a monitoring tool for that territory. Depending on the scope and depth of the analysis outline, LDT can either provide the final information required or can be the first step to evaluate more detailed processes related to biodiversity, resource planning, ecosystem services among others. The principles and methods of landscape ecology have been used by landscape planners and architects in order to preserve, restore and enhance biological diversity (Collinge, 1996) and LDT can help in that mission.

Case study

From 1990 to 2006, both an amount decrement and a NP increment of agro-forestry areas occurred, at the study area scale, suggesting the landscape transformation was characterised as fragmentation due to area loss. This conclusion reflects an oversimplification of the transformations present in the territory because such an approach hides several dynamics, occurring at the same time and in the same landscape that hardly manifest at large scales. This happens because the ultimate result will always be a combination of a value of area change and a value of NP change, for the entire landscape. A multiscale approach is needed to diminish the generalisation and obtain more detailed results. When we look at study area global values, the expected transformation processes are: fragmentation by loss, NP decrement by amount loss, aggregation by amount gain or NP increment by amount gain. This leaves out the raw gain and loss, which are only traceable at smaller scales as they usually concern single patches. If a patch expands or compresses, but not enough to link it to or split it from another patch, this is a composition-related process without relevance in the configuration (concerning the metrics we selected). A gain, if big enough, can lead to an aggregation (by gain) and a loss, if big enough, can lead to fragmentation (by loss). To sum up, the use of a single value for an entire landscape or study area is not viable to track the pure gains or losses, because of its aggregative effect.

In this example, we used maps from two dates only. The observed variations between 1990 and 2006 reveal a loss of agro-forestry areas. However, an intermediate map could show us that the agro-forestry areas were even fewer on that date and in 2006 were recovering. The values would be right but the trend would not. It is always possible to complete or corroborate incomplete data with auxiliary information. For instance, the agro-forestry areas decrement in the Iberian Peninsula in the last decades (Godinho et al., 2016; Plieninger, 2006) is a well-known phenomenon that helps to corroborate our results.

In order to identify dynamics at a local level, it is possible to use analytical units such as squares, hexagons or even irregular forms, if the task demands it. This can be understood as a type of spatial generalisation with the purpose of better representing the data for a specific analytical purpose. Data are often collapsed and aggregated in order to make it more workable, to gain understanding of the phenomenon in question and to uncover patterns masked by the noise usually found in observations. Although the use of analytical units is neither risk-free nor always recommended, it is frequently used (e.g. Cumming & Vernier, 2002) and we think it deserves to be considered as it can be useful to some extent. In our opinion, the use of analytical units represents an important extension to the method proposed by Bogaert et al. (2004). The form and size of the analytical units can be defined by several criteria such as operational (e.g. other data or variables are available in a defined format and a consistent approach is needed or preferential), ecological (e.g. the unit must be big enough to accommodate the phenomena under study), relationship to land management planning units or ownership.

The analysis based on squares reinforced what was revealed by the overall values for the entire study area, a trend of agro-forestry amount loss and a NP increment. Based on the raw values for the whole landscape, we could only speculate this combination would result in fragmentation due to loss.

Through the use of squares, we can achieve more detailed results because we can identify not only fragmentation by loss (dynamic I) but also NP decrement by loss (dynamic H) and pure loss (dynamic E), not considering the squares with gain. There were 92 squares with loss, 45 with NP decrement by loss and 63 with fragmentation by loss (Figure 4(d)). NP decrement by loss and fragmentation by loss are more likely to occur in the presence of greater losses. The larger number of squares with loss (dynamic E) when compared with those of dynamic H and I indicate that in most squares the losses were small. It is more likely for small landscape changes to take place than big ones, thus it is expected to find fewer squares showing big changes and more squares showing small changes. This pattern is identifiable, although with some exceptions, in the different size classes represented in Figure 4(c).

Using Py, Long et al. (2010) were able to demonstrate its ability to differentiate between landscapes primarily experiencing forest loss, concurrent forest loss and configurational change, and those primarily experiencing configurational change. In their study, significant forest loss occurred more frequently than configurational change. Our results are similar in the sense that we detected pure amount changes, a variety of configuration changes originated by amount changes but no strictly geometric transformations.

The case study is demonstrative and does not involve an actual deep analysis of the agro-forestry areas or the drivers that are influencing them. An actual study could use the information provided by the LDT and consider the dynamics locations and probable evolutions, study them, relate them with other variables of interest and come up with valuable insights and practical recommendations.

Conclusions

Future work and improvements

LDT is neither a final nor a closed tool. On the contrary, it is a tool in progress and allows users to make their own modifications. Future works can either be theoretically based or experimentally driven. Theoretical improvements can include the adoption of new paradigms, other ecological concepts or, more likely, fitting the LDT approach in broader analytical contexts. Through practical experiments it will be possible to evaluate the MAUP effects, to test operational specifications like the minimum mapping unit and to include different metrics. Simple metrics that landscape managers will find easy to understand and implement may be more useful but more complex and sophisticated metrics can also be used for quality and accuracy reasons. For instance, after comparing several metrics according to a set of criteria, Jaeger (2000) concluded that the NP should be replaced by the 'effective mesh number' and the 'average patch size' (not the area, though related) should give place to the 'effective mesh size'. Another contribution to reduce the confounding effects of habitat abundance in configuration measurements was made by Wang and Cumming (2009) by introducing normalised versions of three widely used configuration metrics (core area, inter-patch distance and mean patch shape) that can detect and discriminate between positive and negative effects of habitat fragmentation independent of initial habitat abundance. Expected progression can come from new versions of LDT integrating findings and proposals of other authors like the previously mentioned or additional recommendations to include in the procedure. A slightly adapted version of the LDT is being used to assess the spatiotemporal dynamics of the *montado* land cover in South Portugal at large scales, as well as for testing for MAUP effects. The transition of the LDT as a protocol to an ArcGIS toolbox is an obvious step to take that should facilitate its usage and consequently increase its reproducibility.

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References

Bogaert, J., Ceulemans, R., & Salvador-Van Eysenrode, D. (2004). Decision tree algorithm for detection of spatial processes in landscape transformation. *Environmental Management, 33*, 62–73. doi:10.1007/s00267-003-0027-0

Caetano, M., Nunes, V., & Nunes, A. (2009). *CORINE land cover 2006 for continental Portugal, technical report*. Lisbon: Instituto Geográfico Português.

Collinge, S. K. (1996). Ecological consequences of habitat fragmentation: Implications for landscape architecture and planning. *Landscape and Urban Planning, 36*, 59–77. doi:10.1016/S0169-2046(96)00341-6

Cumming, S., & Vernier, P. (2002). Statistical models of landscape pattern metrics, with applications to regional scale dynamic forest simulations. *Landscape Ecology, 17*, 433–444. doi:10.1023/A:1021261815066

Cushman, S. A., McGarigal, K., & Neel, M. C. (2008). Parsimony in landscape metrics: Strength, universality, and consistency. *Ecological Indicators, 8*, 691–703. doi:10.1016/j.ecolind.2007.12.002

Fahrig, L. (1997). Relative effects of habitat loss and fragmentation on population extinction. *The Journal of Wildlife Management, 61*, 603–610. doi:10.2307/3802168

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution and Systematics, 34*, 487–515. doi:10.1146/annurev.ecolsys.34.011802.132419

Fazey, I., Fischer, J., & Lindenmayer, D. B. (2005). What do conservation biologists publish? *Biological Conservation, 124*, 63–73. doi:10.1016/j.biocon.2005.01.013

Fischer, J., & Lindenmayer, D. B. (2007). Landscape modification and habitat fragmentation: A synthesis. *Global Ecology Biogeography, 16*, 265–280. doi:10.1111/j.1466-8238.2006.00287.x

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., … Snyder, P. K. (2005). Global consequences of land use. *Science, 309*, 570–574. doi:10.1126/science.1111772

Forman, R. T. T. (1995). *Land mosaics—The ecology of landscapes and regions*. Cambridge: Cambridge University Press.

Gehlke, C. E., & Biehl, K. (1934). Certain effects of grouping upon the size of the correlation coefficient in census tract material. *Journal of the American Statistical Association, 29*, 169–170. doi:10.2307/2277827

Godinho, S., Guiomar, N., Machado, R., Santos, P., Sá-Sousa, P., Fernandes, J. P., … Pinto-Correia, T. (2016). Assessment of environment, land management, and spatial variables on recent changes in montado land cover in southern Portugal. *Agroforestry Systems, 90*, 177–192. doi:10.1007/s10457-014-9757-7

Haines-Young, R., & Chopping, M. (1996). Quantifying landscape structure: A review of landscape indices and their application to forested landscapes. *Progress in Physical Geography, 20*, 418–445.

Jaeger, J. A. G. (2000). Landscape division, splitting index, and effective mesh size: New measures of landscape fragmentation. *Landscape Ecology, 15*, 115–130. doi:10.1023/A:1008129329289

Jaeger, J. A. G., Madrinan, L. F., Soukup, T., Schwick, C., & Kienast, F. (2011). *Landscape fragmentation in Europe* (EEA Report 2/2011). Copenhagen: European Environmental Agency.

Johnstone, C., Lill, A., & Reina, R. D. (2014). Habitat loss, fragmentation and degradation effects on small mammals: Analysis with conditional inference tree statistical modelling. *Biological Conservation, 176*, 80–98. doi:10.1016/j.biocon.2014.04.025

Leitão, A. B., Miller, J., Ahern, J., & McGarigal, K. (2006). *Measuring landscapes: A planner's handbook*. Washington, DC: Island Press.

Lindenmayer, D. B., & Fischer, J. (2007). Tackling the habitat fragmentation panchreston. *Trends in Ecology and Evolution, 22*, 127–132. doi:10.1016/j.tree.2006.11.006

Long, J., Nelson, T., & Wulder, M. (2010). Characterizing forest fragmentation: Distinguishing change in composition from configuration. *Applied Geography, 30*, 426–435. doi:10.1016/j.apgeog.2009.12.002

McGarigal, K., Cushman, S. A., & Ene, E. (2012). *FRAGSTATS v4: Spatial pattern analysis program for categorical and continuous maps*. Amherst: Computer software program produced by the authors at the University of Massachusetts. Retrieved from http://www.umass.edu/landeco/research/fragstats/fragstats.html

Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being: Synthesis*. Washington, DC: World Resources Institute.

Mitchell, M. S., & Powell, R. A. (2003). Linking fitness landscapes with the behavior and distribution of animals. In J. A. Bissonette & I. Storch (Eds.), *Landscape ecology and resource management—Linking theory with practice* (pp. 93–124). Washington, DC: Island Press.

Openshaw, S. (1984). *The modifiable areal unit problem*. Norwich: Geobooks.

Plećaš, M., Gagić, V., Janković, M., Petrović-Obradović, O., Kavallieratos, N. G., Tomanović, Ž., … Ćetković, A. (2014). Landscape composition and configuration influence cereal aphid-parasitoid-hyperparasitoid interactions and biological control differentially across years. *Agriculture, Ecosystems and Environment, 183*, 1–10. doi:10.1016/j.agee.2013.10.016

Plieninger, T. (2006). Habitat loss, fragmentation, and alteration: Quantifying the impact of land-use changes on a Spanish dehesa landscape by use of aerial photography and GIS. *Landscape Ecology, 21*, 91–105. doi:10.1007/s10980-005-8294-1

Sauder, J. D., & Rachlow, J. L. (2014). Both forest composition and configuration influence landscape-scale habitat selection by fishers (*Pekania pennati*) in mixed coniferous forests of the Northern Rocky Mountains. *Forest Ecology and Management, 314*, 75–84. doi:10.1016/j.foreco.2013.11.029

Saunders, D. A., Hobbs, R. J., & Margules, C. R. (1991). Biological consequences of ecosystem fragmentation: A review. *Conservation Biology, 5*, 18–32. doi:10.1111/j.1523-1739.1991.tb00384.x

Steckel, J., Westphal, C., Peters, M. K., Bellach, M., Rothenwoehrer, C., Erasmi, S., … Steffan-Dewenter, I. (2014). Landscape composition and configuration differently affect trap-nesting bees, wasps and their antagonists. *Biological Conservation, 172*, 56–64. doi:10.1016/j.biocon.2014.02.015

St-Laurent, M.-H., Dussault, C., Ferron, J., & Gagnon, R. (2009). Dissecting habitat loss and fragmentation effects following logging in boreal forest: Conservation perspectives from landscape simulations. *Biological Conservation, 142*, 2240–2249. doi:10.1016/j.biocon.2009.04.025

Turner, M. (1989). Landscape ecology: The effect of pattern on process. *Annual Review of Ecology and Systematics, 20*, 171–197. doi:10.1146/annurev.es.20.110189.001131

Waller, L. A., & Gotway, C. A. (2004). *Applied spatial statistics for public health data*. New York, NY: Wiley.

Wang, X. L., & Cumming, S. G. (2009). Modeling configuration dynamics of harvested forest landscapes in the Canadian boreal plains. *Landscape Ecology, 24*, 229–241.

Wiens, J. A. (1996). Wildlife in patchy environments: Metapopulations, mosaics and management. In D. R. McCullough (Ed.), *Metapopulations and wildlife conservation* (pp. 53–84). Washington, DC: Island Press.

Wilcove, D. S., McLellan, C. H., & Dobson, A. P. (1986). Habitat fragmentation in the temperate. In M. E. Soulé (Ed.), *Conservation biology* (pp. 237–256). Sunderland, MA: Sinauer Associates.

ELSEVIER

Position Paper

LDTtool: A toolbox to assess landscape dynamics

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ABSTRACT

The "LDTtool" python toolbox is introduced. This ArcGIS toolbox can be applied to assess land cover or land use changes between two or three moments, considering landscape composition and configuration. The development of LDTtool occurs in a time when the studies on landscape dynamics looking at amount and configuration changes are becoming mainstream. The use of analytical units, whether regular polygons like squares or irregular like districts, allows to study at broad as well as at detailed scales, depending on the base maps spatial resolution and intrinsic quality. The ultimate goal of the toolbox is to assign each analytical unit to a type of dynamic. The paper shows how to operate the toolbox and provides a case study regarding the olive groves dynamics in Portugal in the period 1990–2018.

1. Introduction

Significant and/or quick landscape changes often disrupts ecosystem functioning interfering with ecological processes and thus jeopardizing their capacity to provide vital services for population (Millennium Ecosystem Assessment, 2005). Substantial landscape changes have such impact on biodiversity that habitat loss and fragmentation are often pointed as one of the major causes of biodiversity loss (Jaeger et al., 2011) in some regions and therefore regarded as a central issue in conservation biology (Saunders et al., 1991; Wiens, 1996). Since landscape patterns influence ecological processes (pattern-process relationship; McGarigal et al., 2012; Turner, 1990), landscape metrics became popular among the landscape ecologists.

To interpret landscapes as discrete patches of different land cover classes (Forman, 1995), there are two main types of changes to consider: composition and configuration changes. The former relates to what constitutes the landscape and its amounts (e.g. land cover types and how

much) – *What exists in the landscape*. The latter has to do with the shape and location of the elements – *How it is distributed*. Although both types of changes co-occur, interact and even depend on each other (Lindenmayer and Fischer, 2007), the separation of their independent effects is necessary to better understand the impacts. Looking at a single measure of change without separating the contribution of the composition and configuration make studies difficult and sometimes impossible to compare (Fahrig, 2017, 2003). An increasing number of studies has been considering the importance of distinguishing the different effects of amount and geometric changes in landscape analysis (e.g. Johnstone et al., 2014; Plećaš et al., 2014; Sauder and Rachlow, 2014; Steckel et al., 2014).

Many metrics, methods and software with distinct specificities have been developed to measure landscape characteristics. Some examples of well-known software are FRAGSTATS (McGarigal et al., 2012), Conefor Sensinode 2.2 (Saura and Torné, 2009), Patch Analyst (Rempel et al., 2012), Circuitscape (McRae et al., 2013), Graphab (Foltête et al., 2012)

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and Land-metrics DIY (Zaragozí et al., 2012). Overall, landscape study does not seem to lack tools but new ones keep being developed in order to improve the existing analytical capabilities (e.g. landscapemetrics by Hesselbarth et al. (2019); gDefrag by Mestre et al. (2019) For a broader notion of the quantity and variety of landscape analytical software existent, especially free and open source, we recommend the works by Steiniger and Hay (2009) and Steiniger and Hunter (2013).

The focus on accounting for both amount and geometric changes is the foundation of the Landscape Dynamic Typology (LDT) method (Machado et al., 2018). LDT is a set of "Types of Dynamics" (ToD) that can occur in a binary landscape, obtained via combination of two metrics that can increase, decrease or remain the same in a period between two dates: *Area* to represent composition and *Number of Patches (NP)* to assess configuration (Table 1).

LDT application can be straightforward but it can also require a

Table 1
Landscape dynamic types (adapted from Machado et al., 2018).

Type	If	and	Designation	Graphic Representation
A	$\Delta A = 0$	$\Delta NP = 0$	No change	$t=0$ → $t=1$
B	$\Delta A = 0$	$\Delta NP > 0$	Fragmentation *per se*	$t=0$ → $t=1$
C	$\Delta A = 0$	$\Delta NP < 0$	Aggregation *per se*	$t=0$ → $t=1$
D	$\Delta A > 0$	$\Delta NP = 0$	Gain	$t=0$ → $t=1$
E	$\Delta A < 0$	$\Delta NP = 0$	Loss	$t=0$ → $t=1$
F	$\Delta A > 0$	$\Delta NP > 0$	NP increment by gain	$t=0$ → $t=1$
G	$\Delta A > 0$	$\Delta NP < 0$	Aggregation by gain (NP decrement by gain)	$t=0$ → $t=1$
H	$\Delta A < 0$	$\Delta NP < 0$	NP decrement by loss	$t=0$ → $t=1$
I	$\Delta A < 0$	$\Delta NP > 0$	Fragmentation by loss (NP increment by loss)	$t=0$ → $t=1$

(A) If there is no change in an area or in the number of patches (NP) we assume that landscape (or the analytical unit extent) did not change; (B) If the area remained the same but the NP increased, it means a fragmentation occurred; (C) If the area remained the same but NP decreased, then an aggregation took place; (D) If the area increased and the NP is equal, it represents a gain of area; (E) If the area decreased and the NP did not change, there is a loss of area; (F) If both area and NP increased, it led to new patch creation; (G) If the area increased and NP decreased, an aggregation occurred due to area gain; (H) If both area and NP decreased, a patch decrement occurred due to area loss; (I) If area decreased and the NP increased, it means that fragmentation occurred due to area loss.

heavy workload and for that reason, the transition of the original protocol to a software piece is regarded as essential to increase its adoption (Machado et al., 2018). In this paper we introduce the "LDTtool", an ArcGIS (www.esri.com) toolbox designed to facilitate and automate the application of the LDT method. We provide a comprehensive description of the toolbox as well as an illustrative case study concerning the olive grove dynamics that have been occurring in southern Portugal.

2. LDTtool

The LDTtool is a python-based add-on ArcGIS toolbox operational in ArcCatalog and ArcMap as well as in ArcGIS Pro. It was developed using the 10.6 version and updates to future versions will be assured in order to keep the toolbox useable.

It comprises five tools (Fig. 1). The first four tools (Landscape Dynamic Types) run the core LDT steps to calculate the dynamics occurred in the landscape. The 1.1 (2 M (Squares)) uses two moments of analysis and regular squares as analytical units. The 1.2. (3 M (Squares)), uses three moments of analysis and squares. The 1.3. (2 M (Districts) uses two moments of analysis and districts as analytical units. The 1.4 (3 M (Districts)) uses three moments of analysis and districts.

Finally, the "2 – Forecast" calculates a hypothetical scenario assuming the ongoing trends will persist. The forecast tool considers how the types of dynamics can evolve from one to the other. For instance, in the presence of area gain, the ToD G (aggregation by gain) will continue eventually until it reaches a 'total cover' situation. In the opposite direction, in a situation of area loss, the ToD H (NP decrement by loss) is going to a 'no cover' scenario when the last patch disappears. For more detailed explanation see Machado et al. (2018).

LDT works with binary landscapes and thus landscape Feature Classes must contain only one class with the polygons, that represent patches, under study (the habitat, the land cover category, etc.).

Relevant recommendations to avoid possible errors or malfunctioning are: (i) using the same coordinate system in the data frame and all the input elements; (ii) using feature classes (requires a geodatabase) instead of shapefiles and (iii) delete unnecessary attribute fields.
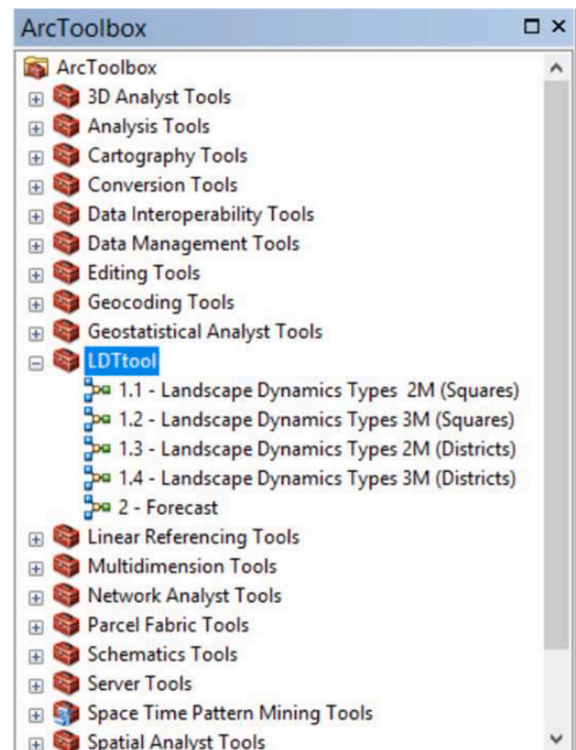

Fig. 1. LDTtool structure in ArcToolbox.

The input's quality also affects the overall quality of the analysis. For that reason, spatial and temporal resolution should be adequate to assess the phenomena under study and therefore reproduce accurately the landscape evolution.

2.1. Preliminary steps

The following preliminary steps are essential to ensure the toolbox correct functioning.

1. Create a File Geodatabase.
2. Make it the default Geodatabase (while creating it or after in: *Menu File, Map Document Properties, Default Geodatabase* (select).
3. Import the feature classes into the Geodatabase. The feature classes should be the landscape moment 1, landscape moment 2 and landscape moment 3 (optional). Regarding the analytical units, the feature classes are the study area boundary (for squares) or districts.
4. Add the "LDTtool" toolbox to ArcToolbox.
5. Select the Geodatabase as "Current workspace" and "Scratch workspace in *Geoprocessing menu, Environments, Workspace* (select).
6. Select the Geodatabase as "Current workspace" and "Scratch workspace" in the Arctoolbox *Environments; Workspace.* (select).
7. Confirm the paths are correct within each tool by Right-clicking; *Properties*; *Environments*; check the *Workspace* box*, Values* button.

2.2. Inputs and settings

The inputs and settings required to run the tools are the following: Tool 1 – Landscape Dynamics Types.

- *Study Area Polygon*: Polygonal feature class containing the study area boundaries.
- *Districts:* Polygonal feature class containing the districts boundaries.
- *Landscape Moment 1*: Polygonal feature class of the landscape in moment 1.
- *Landscape Moment 2*: Polygonal feature class of the landscape in moment 2.
- *Landscape Moment 3*: Polygonal feature class of the landscape in moment 3.
- *Squares width and height (meters):* Analytic square size.
- *Keep patches equal or larger than (square meters)*: Minimum patch size to be analysed.
- *Output Feature Class*: Name and path of the output file.

    Tool 2 - Forecast.

- *Landscape to forecast (Output of tool 1)*

3. The LDTtool demonstration

In order to demonstrate the LDTtool implementation and outputs, we use the tool 1.1 and therefore there are two moments of analysis and squares are used as analytical units. We present an illustrative example of the olive grove dynamics occurred in Portugal between 1990 and 2018.

3.1. Background

Traditional olive groves are a characteristic element of Mediterranean landscapes. Besides its historical and cultural value, traditional groves often host many species making them relevant in terms of biodiversity. In Portugal, the quest for olives and olive oil self-sufficiency first, and exports increments after, led to a landscape change in the last years. New high-yielding intensive (and super-intensive) groves are fast expanding, often replacing biodiversity-rich but low-yielding traditional ones (Morgado et al., 2020). Not only

traditional olive groves but also other land uses such as "non-irrigated arable land" and "permanently irrigated land", have been transformed in large-scale plantations, often with high levels of intensification.

In time, populations began to complain about several problems regarding agrochemicals among other issues. Regarding biodiversity, besides the initial land-cover change that represents a severe habitat loss for many species, it has recently been revealed that the mechanical olives harvesting at night leads to mass bird mortality (Silva and Mata, 2019). At the landscape scale there is a marked homogenization of the territory provoked by large-scale plantations. It is urgent to assure that modern groves, essential to achieve high yield production, minimize their negative environmental impacts while simultaneously the traditional groves are promoted. This might be achieved through both political and/or market mechanisms. For instance, Moreira et al. (2019) suggested new labels should be created to provide consumers with details about the grove from which the product was sourced.

3.2. Preliminary steps

A File Geodatabase was built using ArcCatalog and the feature classes were imported into it (see 2.2 Inputs and Settings subsection). Next, a blank ArcMap project was created and both the default geodatabase and the scratch workspace were pointed to the built geodatabase. Then the "LDTtool" toolbox was added to the ArcToolbox and the feature classes were loaded into the project.

3.3. Inputs and settings

We used the continental Portuguese Administrative Boundaries Official Map (2018 version) as study area and the Olive Groves of 1990 and 2018 extracted from the CORINE Land Cover maps. The analytical square size was defined as 10 km and the minimum size patch (to discard meaningless polygons resulting from intersect operations) as 50,000 m^2. It is important the chosen dimensions are appropriated relative to both the base maps and the phenomena under study. Since CLC uses a minimum size patch of 25 ha (250000 m^2), selecting smaller minimum size patches does not to improve the spatial resolution. Also, we knew beforehand the landscape changes in Portugal involving groves are mostly due to large scale plantations, often larger than 50,000 m^2.

The output feature class was named "LDT_OliveGroves" (Fig. 2a). This feature class was then used as input for "Tool 2 – Forecast" (Fig. 2b).

3.3.1. Square size selection

The tool allows the use of districts and squares as analytical units. Although the pros and cons are common to both types of analytical units (see subsection "3.5. Results and discussion" for more on the Modifiable Areal Unit Problem - MAUP), the districts are usually used for some specific reason and often exempted from further justification while the artificial sampling schemes such as regular grids tend to be more scrutinized. For example, a county boundary is as artificial for some natural processes as a square or an hexagon but it is commonly accepted that data has to be aggregated by county our any other administrative or statistical unit. In analytical terms the repercussions may be similar but when using an artificial sampling grid an extra attention is required to make sure the sampling scheme itself is not adulterating the raw data. Since there is not a single solution for this problem and there are a multitude of contexts and applications, users often have to find a way to assess quality and establish tolerance levels. Regular areas like squares can be used for no particular reason but many times are used because there are many associated data available in that format, thus allowing variable extraction for posterior analysis. One important aspect to consider is how well the sampling scheme preserves the integrity of the base data. For instance, in our case study, a single patch can be recognized by the software as being multiple patches because it crosses several squares (or districts). Since the final calculations are based on the analytical unit and not on the patches themselves, the same patch
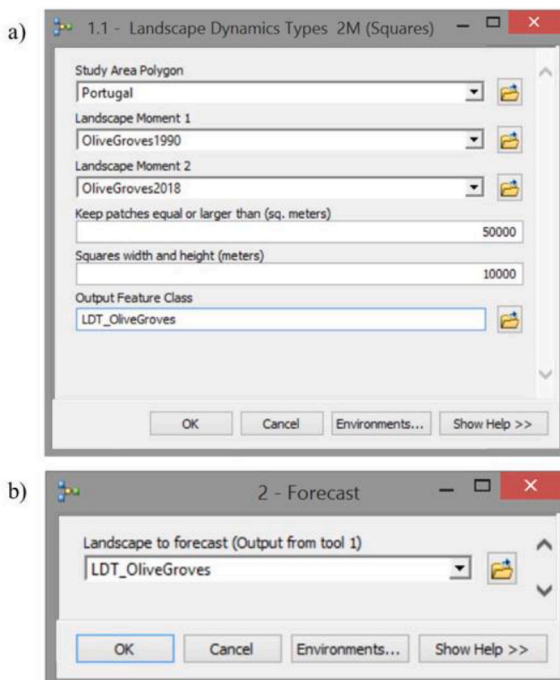
Fig. 2. Filled dialog boxes. a) Tool 1.1 - Landscape dynamics Types 2 M (Squares); b) Tool 2 - Forecast.

can be counted multiple times. Ultimately, the only way to solve this problem was to use the study area boundaries and discard the use of analytical units. The downside of such an option is the complete loss of spatial resolution and its associated benefits. In our view, a good compromise would be a grid size that minimizes patch intersection and still offers an adequate spatial resolution to assess the phenomenon being studied. To what NP are concerned, olive groves maps show 1613 patches in 1990 and 1974 in 2018. After intersecting the grid composed by 1011 100 km^2 squares, the NP was 2220 and 2685, respectively. This represents 607 artificial extra patches for 1990 and 711 for 2018. The intersections per square were 2,2 for 1990 and 2,7 for 2018. To improve the spatial resolution requires smaller squares that would make these numbers worse. On the opposite direction, to improve these numbers we would have to enlarge the squares and lose spatial resolution.

As to patch size, the average olive grove was 1,72 km^2 in 1990 and 1,76 km^2 in 2018. Thus, a 100 km^2 square would be large enough to fit several patches and to minimize the original patches crossing several squares. Looking at the extremes of the patch size distribution, only one patch in 1990 and 2018 is larger than 100 km^2. This suggests that the grid size is not too small. It could be larger but the larger the square is, the more patches it can fit and that increases the risk of hiding some ToD by losing spatial resolution. Overall, these values seemed acceptable for the task in hands but could the output be too influenced by the square size? To answer this question we reran the tool using 9 km, 10 km, 11 km and 12 km squares that differ 10% and 20% from the original 10 km grid. This allowed us to assess the MAUP's scale effect. Having different sizes, the grids are distinct, the squares are not in the same place, and with that we were also assessing the MAUP's context or zoning effect. With the 10 kmX10km squares as a starting point, we used R software (R Core Team, 2018) to run Chi-Squared tests against all the other grid sizes, based on the ToD counts. The results show the ToD counts display no significant differences and thus are considered similar. A complementary visual analysis of the ToD spatial distribution also reveals consistency among different square sizes, albeit some expected variations. The Chi-Squared test results and the maps are provided as supplementary material.

3.4. Outputs

The final product is a feature class with the study area divided by squares, each one with the metrics calculated for both dates, their variation and assigned to a ToD (Figs. 3 and 4).

3.5. Results and discussion

The aim of this particular case study was to illustrate how to use the LDTtool and show the type of outputs it produces. The analysis was a broad one and did not intend to explain exhaustively the reality behind the olive groves increment in Portugal, which is a complex issue and would require more and different data with higher spatial resolution, possibly thematic resolution (distinguish between traditional and modern olive groves) and temporal resolution (use intermediary dates to better identify the paces in the land cover changes).

Although other drivers may be responsible for local changes occurring all over the country, it is hard to ignore the process of rapid agricultural intensification going on in the Alentejo Region (Southern Portugal) and its relevance to the landscape transformation. The traditional extensive, multi-functional agricultural systems, adapted to the Mediterranean climate are being rapidly replaced by more intensive and irrigated cultures. This was made possible due to public investment in the Alqueva dam and its integrated irrigation system, together with national and EU agricultural policies (Silveira et al., 2018). As a result, large scale intensive and super-intensive olive groves have been implemented in the last years in newly irrigated areas.

According to CLC (CHA00, CHA06, CHA12 and CHA1218) the land covers that have been replaced the most by olive groves are "non-irrigated arable land" and "permanently irrigated land", presumably land that was not irrigated before but has now access to water and land that was already irrigated but was used for other cultures. In opposition, some existing olive groves were replaced by, mostly, "non-irrigated arable land" and "transitional woodland-shrub", what can reflect the low income obtained by traditional olive groves that were abandoned.

Comparing the 1990 and 2018 feature classes we verify there have been an olive grove area increment of 692,84 km^2 and a number of patches increment of 361. These overall values suggest the "Type of Dynamic F–NP increment by gain" has been the main change pattern taking place at the study area. The results based on the 100 km^2 grid provide more detailed results. The number of squares assigned to each ToD at this spatial resolution is present in Fig. 5a).

The "ToD F–NP increment by gain" was the most represented ToD with a total of 227 squares which is in line with the global trend previously mentioned. ToD F represents new olive tree plantations not adjacent to existing ones. The magnitude of ToD F reflects how vigorous the olive grove implementation in Portugal has been. Nevertheless, there were some local area losses, presumably traditional and less profitable groves. The second most identified ToD was the "H–NP decrement by loss", present in 101 squares. It shows squares where the main dynamic was the shift from olive groves into other land covers. Looking at the number of squares it seems quite significant with almost half of the ToD F (101 vs 227). However, the relation does not stand when we compare olive grove area variation as ToD F involved an area gain of 817,57 km^2 while ToD H involved an area loss of 159,61 km^2 (Fig. 5b).

The other two ToD with variation in both configuration and composition are "G – Aggregation by gain" that reflects situations where new plantations or expansion of existing ones originated the fusion of patches (groves), and "I – Fragmentation by loss", where the loss of area provoked the division of patches. ToD G was identified in only 16 squares but totalled an area gain of 143,82 km^2, while ToD I was identified in 55 squares and implied an area loss of 118,05 km^2. Focusing on the ToDs that reflected only composition and had no influence in the NP, "D – Gain" was found in 49 squares and involved an area gain of 57,18 km^2, while "E – Loss" occurred in 71 squares with a

Fig. 3. Output attribute table. a) Output of Tool 1.1 - Landscape dynamics Types 2 M (Squares); b) Field originated by Tool 2 – Forecast.
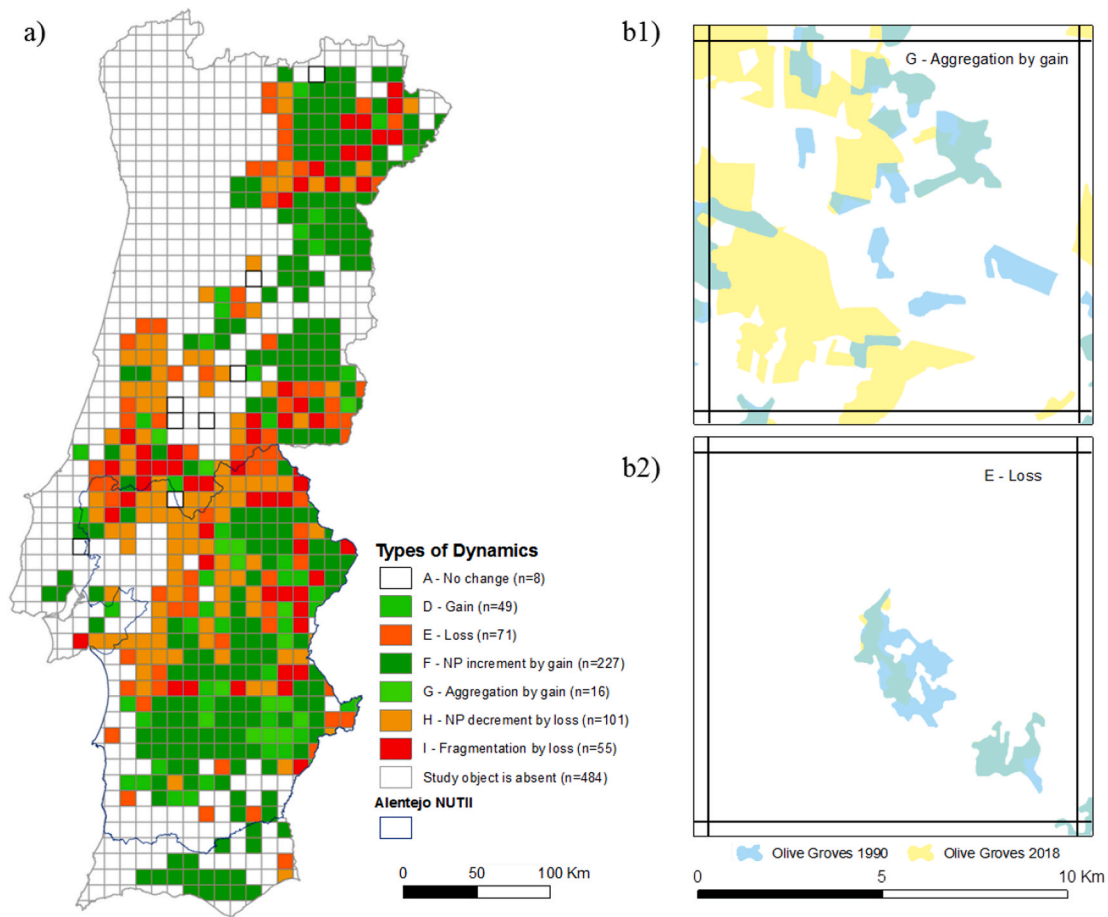

Fig. 4. Olive Groves in Portugal (1990–2018). a) Map produced by "Tool 1 - Landscape Dynamics Types 2 M (Squares)" showing Continental Portugal divided by 10 × 10 Km squares with assigned Types of Dynamics (ToD). b1) Detailed view of a square showing TOD G - Aggregation by gain. b2) Detailed view of a square showing ToD E − Loss.

total loss of 47,83 km². No squares were assigned to the ToD B or C, that represent pure geometric changes (without amount variation) and thus do not occur often.

In summary, there were many changes in olive groves in Portugal between 1990 and 2018. Overall, the area increased despite some local losses. Most of the amount changes had implications on the olive groves configuration (shape and spatial distribution). In a situation (or scenario) of olive groves substantial area increment, all the ToD related to area gain may contribute to landscape homogenization. Obviously, the area amount turned into olive groves is the main thing responsible for that contribution but considering a fixed amount, the way it is added to

the landscape is also relevant. While pure gain, meaning the existing patches were expanded (ToD D – Gain), and the presence of new patches (ToD F–NP increment by gain) have influence mainly at the landscape granularity level, the fusion of patches (ToD G – Aggregation by gain) contributes directly to the landscape spatial homogenization. Also related to this, particularly if considering biodiversity issues, it is fundamental to verify the starting point of each square as a given amount of area change may have different meanings and effects. In other words, similar areal increments may have different impacts if the square had none of that cover, or if the square already had much of it. The LDT forecast tool should be used with caution as its purpose is to support the
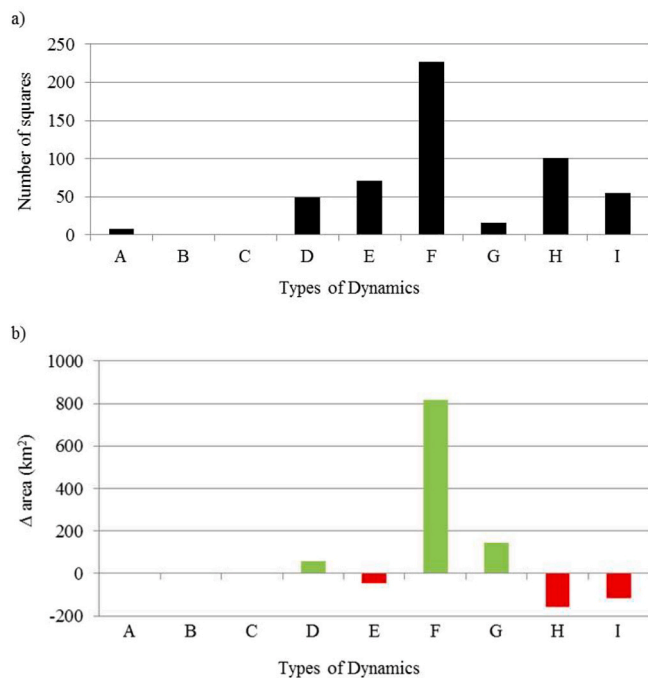
Fig. 5. a) Number of squares assigned to each Type of Dynamics; b) Olive grove area variation associated to each Type of Dynamics.

thinking process of how the landscape may look like in the future and not to make accurate predictions. Based on the LDT forecast model, we can see that if there is an increasing trend, in time, the existing patches of a given land cover may expand so that they bridge the gaps between them and merge together (ToD D evolves to ToD G). New patches that are installed in the landscape may follow the same path (ToD F is followed by ToD D which evolves to ToD G). Such analysis act as a diagnosis and projects a hypothetical future based on the current trend. Nevertheless, we must advise this tool is better suited for situations where natural processes are in place without management or with little human interference (e.g. vegetation recovery, afforestation, etc.). In some cases it could be useful to study different forms of ownership, for instance large landowners who manage intensively and a multitude of small owners who have no interest in management and this consequently leads to abandonment of agricultural use. In the case of planned, highly managed olive groves (or any other agricultural land) the land use changes are mostly dependent of human decision. For instance, the tool shows us the ToD F and ToD D may end up originating a higher number of ToD G squares, which implies a greater spatial homogeneity than we have today. This may end up being true but it is hard to sustain this prediction because the olive groves will not expand naturally via dispersal. Instead they will appear and disappear due to human direct intervention in the territory.

The analytical grid allowed us to identify local dynamics along the study area and visualize them in a map (Fig. 4a). This enhanced capability can be relevant for planning, management and decision making processes. Although there are valuable advantages provided by the use of analytical units, we must highlight the modifiable areal unit problem (MAUP) (Gehlke and Biehl, 1934; Openshaw and Taylor, 1979). The MAUP refers to the fact that the areal units can be set arbitrarily and are modifiable. Results based on aggregated data following areal units can change if the same data are aggregated under different areal units (Waller and Gotway, 2004). Commonly used in population-focused studies, data aggregation is often based on boundaries such as zip codes, census tracts and census block groups (Moon and Farmer, 2001). Regular grids are extensively applied in spatial analysis to avoid bias caused by administrative divisions (Swift et al., 2008) and/or to include

the vast amount of data that is available in that format, such as remote sensing data, biological atlas or simply because there is a preference for working with squares (e.g. UTM) or hexagons as a reference.

MAUP is a long standing issue in geography and spatial analysis that has no optimal solution and has been ameliorated by the researchers according to their study contexts. Among several attempts that were made to deal with the MAUP (Openshaw, 1984), proposed that areal units should match the optimal spatial variance or maximize a given statistic. This is a clear protocol that leaves no doubt as to how the units are designed, but at the same time it is easy to criticize as it allows researchers to design spatial units in order to achieve a preferred result. According to Swift et al. (2008), areal units generated from Voronoi tessellations (Thiessen polygons) can be effective to reduce aggregation bias. The same authors point out that sensitivity analysis can be superior to other methods (e.g. optimizing the size and shape of areal units) since it does not rely on highly accurate measurements of spatial variables. Long et al. (2010) highlighted that MAUP influences the results, and suggested a multiscale approach when implementing their metric 'proportion of landscape displacement from configuration' (Py) as a way to minimize scale-related bias. Butkiewicz et al. (2008) first introduced a geospatial visualization method and later provided a number of enhancements to help lighten the effects of the MAUP (Butkiewicz et al., 2010).

In our example, if the squares had a different size (scale effect) and/ or a different location (context effect), their content in terms of olive groves would also be different. Scale effect assessment would require testing different square sizes and context effect assessment would require changing their location. Although the current LDTtool version does not include such validation instruments, similar operations can be conducted by running the tool using different size grids (scale effect) and feeding the tool with edited grids (rotation, translation, etc.) via tool 1.3. or 1.4. (context effect). We took a similar approach and added a statistical test to assess the outputs (see Supplementary Material). A future validation/quality control procedure to integrate LDTtool is likely to involve an automatization of such steps and will be incorporated in the existing tools or provided as separated analytical tools.

4. Concluding remarks

The newly developed LDTtool can add value to studies in a wide range of topics related to biodiversity conservation, invasive species, ecosystem services and natural resources planning, mainly if land use, land cover or habitat changes are a central topic of the study. After the toolbox functionalities were presented, a demonstration was provided using as an example the olive groves dynamics in Portugal in the period between 1990 and 2018.

4.1. Strengths, limitations and future developments

The LDTtool's strengths and limitations are twofold: those intrinsic to LDT and those related to the software itself. Some of the LDT method's virtues are its simplicity due to the fact of being based on simple spatial metrics and the easy interpretation of the outputs that come in the form of value variations and maps (Machado et al., 2018). LDTtool itself, being introduced as an ArcGIS toolbox is expected to be user-friendly. The toolbox configuration is straightforward and it runs smoothly and fast mostly because it uses the vectorial format. LDTtool is implemented/coded in Python, which is a versatile language widely used in geoprocessing. Additionally the source-code is available and users can extract it and edit it to fit their purposes. The possibility to adjust parameters such as the minimum patch size and the analytical units also add versatility to the process. However, the use of analytical units (regular polygons such as squares or irregular polygons such as districts) does not come without a cost, which in this case is related to MAUP. Variations in how regions are delineated have an influence in how the data is aggregated. Scale has similar implications because local variation

can be lost when aggregated into a larger region (e.g. 20 Km squares instead of 10 Km squares). Therefore, both scale and analytical units must be cautiously selected in order to fit the purpose of the study. Although this is up to the user, we are aware that an automated validation procedure that helps mitigate the MAUP implications would highly improve LDTtool.

Also the use of binary landscapes facilitates the analysis but can be too simplistic to represent reality, mainly in biodiversity studies, particularly those involving habitat suitability or functional connectivity.

The current version is not a final product but rather the first version of a tool that has room to improve and will be updated to assure its usability. LDTtool's current limitations are identified and will be the base for future developments. Some upcoming improvements being considered at the moment are (i) quality (sensibility or robustness) analysis of the final results to minimize the MAUP limitations, (ii) expand the analytical possibilities beyond binary landscapes, and (iii) integration in other GIS platforms.

Software availability

Name: LDTtool.

Availability and cost: LDTtool can be can be freely downloaded from: https://github.com/RDPMachado/LDTtool.

(Includes a README file with description and instructions).

License: GNU GPLv3; Developer: Rui Machado; Contact address: Departamento de Paisagem, Ambiente e Ordenamento, Universidade de Évora, Rua Romão Ramalho, n◦ 59, 7000-671 Évora, Portugal, E-mail: rdpm@uevora.pt.

Year first available: 2020.

Software required: ESRI® ArcMap™ 10.0 and later versions.

Program language: Python.

Program size: 6.97 Mb.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envsoft.2020.104847.

References

Butkiewicz, T., Dou, W., Wartell, Z., Ribarsky, W., Chang, R., 2008. Multi-focused geospatial analysis using probes. IEEE Trans. Visual. Comput. Graph. 14, 1165–1172.

Butkiewicz, T., Meentemeyer, R.K., Shoemaker, D.A., Chang, R., Wartell, Z., Ribarsky, W., 2010. Alleviating the modifiable areal unit problem within probe-based geospatial analyses. Comput. Graph. Forum 29, 923–932. https://doi.org/10.1111/j.1467-8659.2009.01707.x.

Fahrig, L., 2017. Ecological responses to habitat fragmentation per Se. Annu. Rev. Ecol. Evol. Syst. 48 https://doi.org/10.1146/annurev-ecolsys-110316-022612 annurev-ecolsys-110316-022612.

Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. Annu. Rev. Ecol. Evol. Syst. 34, 487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419.

Foltête, J.C., Clauzel, C., Vuidel, G., 2012. A software tool dedicated to the modelling of landscape networks. Environ. Model. Software 38, 316–327. https://doi.org/10.1016/j.envsoft.2012.07.002.

Forman, R.T.T., 1995. Land Mosaics - the Ecology of Landscapes and Regions. Cambridge University Press, Cambridge.

Gehlke, C.E., Biehl, K., 1934. Certain effects of grouping upon the size of the correlation coefficient in census tract material. J. Am. Stat. Assoc. 29, 169–170.

Hesselbarth, M.H.K., Sciaini, M., With, K.A., Wiegand, K., Nowosad, J., 2019. landscapemetrics: an open-source R tool to calculate landscape metrics. Ecography 42, 1648–1657.

Jaeger, J.A.G., Soukup, T., Schwick, C., Madriñán, L.F., Kienast, F., 2011. Landscape Fragmentation in Europe, European Landscape Dynamics: CORINE Land Cover Data. https://doi.org/10.1201/9781315372860.

Lindenmayer, D.B., Fischer, J., 2007. Tackling the habitat fragmentation panchreston. Trends Ecol. Evol. 22, 127–132. https://doi.org/10.1016/j.tree.2006.11.006.

Long, J.A., Nelson, T.A., Wulder, M.A., 2010. Characterizing forest fragmentation: distinguishing change in composition from configuration. Appl. Geogr. 30, 426–435. https://doi.org/10.1016/j.apgeog.2009.12.002.

Machado, R., Godinho, S., Pirnat, J., Neves, N., Santos, P., 2018. Assessment of landscape composition and configuration via spatial metrics combination: conceptual framework proposal and method improvement. Landsc. Res. 43, 652–664. https://doi.org/10.1080/01426397.2017.1336757.

McGarigal, K., Cushman, S.A., Ene, E., 2012. FRAGSTATS v4: spatial pattern analysis program for categorical and continuous maps. In: 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/f.

McRae, B.H., Shah, V.B., Mohapatra, T.K., 2013. Circuitscape 4 user guide. In: The Nature Conservancy. http://www.circuitscape.org.

Mestre, F., Ascensão, F., Barbosa, A.M., 2019. gDefrag: a graph-based tool to help defragmenting landscapes divided by linear infrastructures. Ecol. Model. 392, 1–5. https://doi.org/10.1016/j.ecolmodel.2018.10.012.

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Synthesis. Washington, DC.

Moon, Z.K., Farmer, F.L., 2001. Population density surface: a new approach to an old problem. Soc. Nat. Resour. 14, 39–51. https://doi.org/10.1080/089419201300199545.

Moreira, F., Herrera, J.M., Beja, P., 2019. Making olive oil sustainable. Science 84 365, 873. https://doi.org/10.1126/science.aay7899.

Morgado, R., Santana, J., Porto, M., Sánchez-Oliver, J.S., Reino, L., Herrera, J.M., Rego, F., Beja, P., Moreira, F., 2020. A Mediterranean silent spring? The effects of olive farming intensification on breeding bird communities. Agric. Ecosyst. Environ. 288 https://doi.org/10.1016/j.agee.2019.106694.

Openshaw, S., 1984. The Modifiable Areal Unit Problem. GeoBooks, Norwich, England. https://doi.org/10.1002/9781118526729.ch3.

Openshaw, S., Taylor, P., 1979. A million or so correlation coefficients: three experiments on the modifiable areal unit problem. In: Wrigley, N. (Ed.), Statistical Applications in the Spatial Sciences, pp. 127–144. Pion, London.

R Core Team, 2018. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. URL. https://www.R-project.org/.

Rempel, R.S., Kaukinen, D., Carr, A.P., 2012. Patch Analyst and patch grid. In: Ontario Ministry of Natural Resources. Centre for Northern Forest Ecosystem Research, Thunder Bay, Ontario. Available from: http://www.cnfer.on.ca/SEP/.

Saunders, D.A., Hobbs, R.J., Margules, C.R., 1991. Biological consequences of ecosystem Fragmentation : a review. Conserv. Biol. 5, 18–32.

Saura, S., Torné, J., 2009. Conefor Sensinode 2.2: a software package for quantifying the importance of habitat patches for landscape connectivity. Environ. Model. Software 24, 135–139. https://doi.org/10.1016/j.envsoft.2008.05.005.

Silva, L.P., Mata, V.A., 2019. Stop harvesting olives at night — it kills millions of songbirds. Nature 569.

Silveira, A., Ferrão, J., Muñoz-Rojas, J., Pinto-Correia, T., Guimarães, M.H., Schmidt, L., 2018. The sustainability of agricultural intensification in the early st insights from the olive oil production in Alentejo (Southern Portugal). In: Delicado, A., Domingos, N., de Sousa, L. (Eds.), Changing Societies: Legacies and Challenges. Vol. III. The Diverse Worlds of Sustainability. Imprensa de Ciências Sociais, Lisbon, pp. 247–275. https://doi.org/10.31447/ics9789726715054.10.

Steiniger, S., Hay, G.J., 2009. Free and open source geographic information tools for landscape ecology. Ecol. Inf. 4, 183–195. https://doi.org/10.1016/j.ecoinf.2009.07.004.

Steiniger, S., Hunter, A.J.S., 2013. The 2012 free and open source GIS software map - a guide to facilitate research, development, and adoption. Comput. Environ. Urban Syst. 39, 136–150. https://doi.org/10.1016/j.compenvurbsys.2012.10.003.

Swift, A., Liu, L., Uber, J., 2008. Reducing MAUP bias of correlation statistics between water quality and GI illness. Comput. Environ. Urban Syst. 32, 134–148. https://doi.org/10.1016/j.compenvurbsys.2008.01.002.

Turner, M.G., 1990. Spatial and temporal analysis of landscape patterns. Landsc. Ecol. 4, 21–30.

Waller, L.A., Gotway, C.A., 2004. Applied Spatial Statistics for Public Health Data. John Wiley and Sons, New York.

Wiens, J.A., 1996. Wildlife in patchy environments: metapopulations, mosaics and management. In: McCullough, D.R. (Ed.), Metapopulations and Wildlife Conservation. Island Press, Washington, DC, pp. 53–84.

Zaragozí, B., Belda, A., Linares, J., Martínez-pérez, J.E., Navarro, J.T., Esparza, J., 2012. A free and open source programming library for landscape metrics calculations. Environ. Model. Software 31, 131–140. https://doi.org/10.1016/j.envsoft.2011.10.009.

Supplementary material

We ran Chi-Squared tests to assess MAUP. We tested the results (ToD counts) of the 10kmX10km squares against those of 8km, 9km, 11km and 12km. The results came with the same outcome: Not rejecting the null hypothesis that assumed the values are similar / equivalent (not significantly different).

ToD	Size	
	8	**10**
A	9	8
D	76	49
E	116	71
F	296	227
G	26	16
H	153	101
I	60	55
Absent	803	484

X-squared = 9.2028, df = 7, p-value = 0.2384

ToD	Size	
	9	**10**
A	11	8
D	68	49
E	95	71
F	239	227
G	25	16
H	102	101
I	66	55
Absent	627	484

X-squared = 6.8288, df = 7, p-value = 0.4469

ToD	Size	
	10	**11**
A	8	6
D	49	43
E	71	61
F	227	202
G	16	16
H	101	77
I	55	47
Absent	484	389

X-squared = 1.5007, df = 7, p-value = 0.9823

ToD	Size	
	10	**12**
A	8	7
D	49	34
E	71	39
F	227	174
G	16	15
H	101	74
I	55	49
Absent	484	327

X-squared = 4.8826, df = 7, p-value = 0.6743

Maps showing consistent spatial distribution of ToD assigned to different square sizes. Despite the natural variation in the spatial resolution, the major patterns are still recognizable in all maps.

| 12km | 11km | **10km** | 9km | 8km |
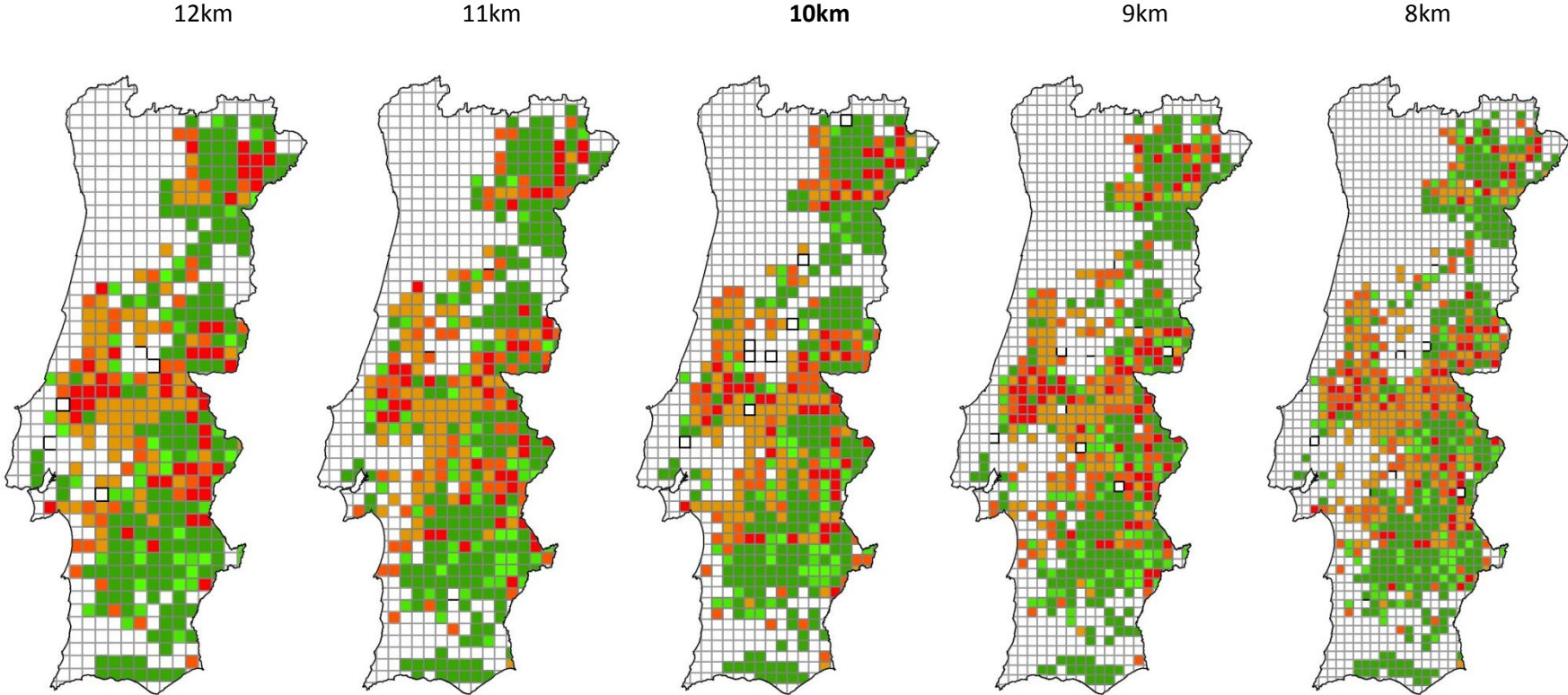
RESEARCH ARTICLE

Supporting the spatial management of invasive alien plants through assessment of landscape dynamics and connectivity

Rui Machado[1,2] , Liliana Neto Duarte[3] , Artur Gil[4,5], Nuno Sousa-Neves[6,7], Janez Pirnat[8], Pedro Santos[9]

Invasive alien species are responsible for several negative impacts worldwide. Managing biological invasions is often difficult and the success rate is quite low, but with good planning it is possible to achieve good results. Besides employing the correct methods and techniques, an overall strategy based on landscape dynamics and expected spatial patterns can be fundamental to achieve success. The decision of where to act can be embedded in a general strategy based on several criteria/goals such as control of large populations, connectivity disruption, and so on. This work focused on *Acacia dealbata* in a Natura 2000 site in Portugal, how the current amount and distribution can affect the spread pattern, and different possible strategies to approach the management. Based on the species dispersal traits, we argue that not only the area but also the perimeter (therefore, the shape) and location of the patches should be considered when fighting the invasion. Three scenarios were designed and compared using the perimeter–area ratio, a landscape dynamics analysis, and a connectivity index. Results show that removing the patches with higher perimeter–area ratio (mostly small satellite patches) would be more impactful than removing the larger patch or removing random intermediary perimeter–area patches first. After this approach based on landscape dynamics, the employment of a connectivity assessment provided an ordered list of patches to remove sequentially. Overall, this approach can be valuable in the early steps of the planning process, supporting better decisions regarding the available resources and contributing to maximize the effectiveness of the action.

Key words: *Acacia dealbata*, invasive alien plants, landscape ecology, restoration, spatial patterns

Implications for Practice

- In a task as challenging and expensive as restoring invaded areas, planners need to know where the scarce resources can be employed more effectively.
- Removing larger patches of invasive plants is not always the most effective way to control the invasion.
- Patch size, shape, and location all contribute to the distribution and dispersal of the invasive plant and should be assessed to learn where a certain amount of work can produce optimal results.

Introduction

Invasive alien species (IAS) rank in the top five direct drivers of change in nature (IPBES 2019), being responsible for several negative impacts (Kumschick et al. 2015; Schindler et al. 2015; Jones 2017; Diagne et al. 2020). Biodiversitywise, IAS are listed among the major indicators of decline (Butchart et al. 2010) and the second most common threat associated with recent extinctions in five major taxonomy groups (plants, amphibians, reptiles, birds, and mammals) (Bellard et al. 2016). Although biological invasions are a major driver of ecosystem degradation, there is no evidence that the rate of alien species introduction is slowing down, and the number of those becoming invasive is even increasing (Pyšek et al. 2020). Biological invasions are a

pervasive component of global change impacting ecosystems in several ways (Simberloff et al. 2013). Invasive alien plants (IAP) can alter basic ecological processes such as nutrient cycling or the change of soil biota (Marchante et al. 2008; Vilà et al. 2011). They often benefit from competitive mechanisms (Levine et al. 2003) or performance-related traits (Van Kleunen et al. 2010), which includes long-lived seed banks (Gioria & Pyšek 2015), disrupting intrinsic interactions among native species (Callaway & Aschehoug 2000).

Management of IAS is frequently difficult and expensive (Diagne et al. 2021), but with proper planning and resources, biological invasions can be managed and mitigated (Pyšek et al. 2020). First, it is important to define "impact" precisely. More than focusing on evident negative impacts caused by non-native species, appropriate management and policies benefit from an understanding of impact in a broader sense, able to consider, e.g. directionality and scale (Jeschke et al. 2014). Second, it is essential to define what "success" is for each intervention prioritizing both the species and the areas (van Wilgen et al. 2012). According to the existing situation and the available resources, the goal can be to control, contain, or even eradicate the species (Simberloff et al. 2013). Control and containment may seem similar concepts, but an instructive distinction is made by Hulme (2006). Control aims to reduce the impact and the abundance of an IAS to an acceptable level, although not necessarily limiting its range. It can be interpreted as managing and trying to minimize an inevitable or a bearable dispersion. On the contrary, containment aims to limit the spread by acting mostly at the periphery of the species range. This makes containment more appropriate for species that disperse slowly and over short distances, while control may become more realistic when dealing with larger peripheries typical of long dispersers with higher expansion rates. Eradication implies eliminating all the individuals and viable propagules of a species within the management unit (Parkes & Panetta 2009). It also requires that the species has not been detected for a period equal to or greater than its seed longevity (Panetta 2007), and therefore it is usually difficult to achieve once the species is established. According to Mack and Lonsdale (2002), the record of eradicating invasive plants consists of "few clear victories, some stalemates, and many defeats." We must accept the possibility that some ecosystems will not completely recover and be aware that the idea of "success" in a restoration project may change over time (Cordell et al. 2016). Even if the natural vegetation is reestablished, the associated fauna is not necessarily recovered. The systems are usually complex, and the emphasis of the monitoring is placed on the return of a few target groups such as plants and do not consider the entire food web that they supported before the invasion (Zedler & Callaway 2000; Gratton & Denno 2005).

Many interventions see the removal of alien vegetation as the final goal (Vosse et al. 2008), but that alone does not guarantee the natives will come back and restore the ecosystem as it was before the invasion (D'Antonio & Meyerson 2002; Harms & Hiebert 2006). In fact, passive restoration approach often fails to avoid reinvasions by the same species or secondary invasions by other species, able to capitalize on the disturbance caused by

the removal operation (Holmes et al. 2020). Furthermore, ecosystem recovery may fail due to the IAP legacy, in the form of reduced biodiversity, massive seed banks, altered soil chemistry, among other factors (Corbin & D'Antonio 2012). Clear examples are provided by Marchante et al. (2008, 2019) regarding the soil chemistry alterations due to invasion by *Acacia longifolia*.

To achieve an actual recovery rather than an IAP removal with ephemeral results, the elimination of the invaders must be accompanied by strategies to overcome their legacies (Konlechner et al. 2015). Such active approaches to restoration are more complex and often involve revegetation with native plants following IAP removal, which requires additional knowledge about the species and the area itself. Aspects such as the soil, the climate, the IAP ecological requirements to occupy habitats, and the dynamics of native plant communities can be critical to the success of the operation (Duarte et al. 2020). Overall, active approaches can be more expensive at the start but tend to deliver better results in the long run (Gaertner et al. 2012).

More resources, mainly financial, allow for more extensive operations and increased effort, such as follow-up treatments, but the restoration should always be carefully planned to prevent poor-quality implementation that results in the need for larger budgets over time (Cheney et al. 2019). Besides the financial issues, the feasibility of the restoration depends on the area itself and on the invasion debt (the delayed spread after the introduction and the escalation of impacts over time) (Pyšek et al. 2020). If the area is one of conservation priority or is surrounded by quality matrix (natural or semi-natural habitats), active restoration may be worthwhile, but if that is not the case, it may not be practicable or reasonable (Gaertner et al. 2012). One way to maximize recovery and reduce costs of active restoration is to prioritize patches with higher natural restoration potential (i.e. spontaneous succession potential). Higher recovery rates may be anticipated in smaller, more distant, and younger patches than in large, less distant patches that are probably older and well established. Also the recolonization by the species that were present before the invasion is probably faster in small than in large species where the seed sources may be remote from the center of the patch (Turner et al. 1994).

A multitude of relevant factors for IAP management are included in comprehensive approaches or procedures introduced by several authors. Higgins et al. (2000) developed a broad conceptual model that includes parameters such as spread patterns, time needed to eradicate the plants, costs of the action, and a number of spatial variables to help define clearing strategies. Krug et al. (2010) tested budget scenarios and the associated efficiency to better prioritize cleaning areas. The method is versatile and can accommodate key factors for the spread of the species under study. Roura-Pascual et al. (2009) based their study on multiple environmental and socioeconomic factors and developed a comprehensive framework for prioritizing areas for managing woody IAPs. Their results for a South Africa fynbos case study highlight the fire-prone nature of the ecosystem and the invasive stands characteristics as relevant features for management. To account for uncertainty in the analytical procedures regarding the management of woody IAPs, Roura-Pascual

et al. (2010) developed a framework for a spatially explicit sensitivity analysis including factors related to fire risk, IAP spread and density, and others. Our work focuses on spatial prioritization of control operations and can produce useful data on spatial patters and connectivity of IAP, to feed broader analytical procedures. It is essential to decide where to employ the efforts considering the available resources and anticipating what the expected outcome will be. In other words, given the available resources, where can the intervention get the maximum return by employing adequate techniques? More precisely, this work addresses the landscape patterns, particularly the abundance and distribution of the invasive species and where its removal could be more beneficial.

The pressure that individual patches suffer from the surrounding matrix is something to consider. In that context, edges are critical because they are the interface between the interior and the exterior. Decreasing the ratio of edges to interiors helps to minimize exposure from external influences. However, when an IAP patch is spreading it successfully exerts its influence in the matrix or the surrounding patches. If the patch gains area by advancing gradually as a front, it uses its edge to pressure the contiguous land-covers. Therefore, the less edge the better, to halt the patch expansion. Area becomes particularly relevant when dealing with species with long-distance dispersal mechanisms because regardless of how well the edge is contained, they can always spread long distances and start new foci. The relative importance of edge length and area may vary according to the species but they are both relevant as they depend on the patches' size and shape. For this reason, the perimeter–area ratio (PAR) may be particularly useful to select which IAP patches to remove (first). Using PAR as the main criterion, the smaller patches are expected to be selected first because they usually display higher PAR values, but larger and irregular patches may also rank in top positions.

It is appealing to remove the larger patches of IAP because they seem more threatening than smaller patches; because it produces more noticeable work; or simply because the impacts do not become evident and problematic until the invaders are well-established and cover large areas (Moody & Mack 1988; Pyšek et al. 2020). Nevertheless, many authors have suggested that removing smaller patches is crucial to slow down the invasion rate and stop the spreading in the future (Campbell 1993). The dispersal mechanism is dependent on the species, but generally, after the introduction, the spreading takes place via a patch that advances as a front (usually large patch[es]) and may be the source of "satellite" populations (smaller patches) (Radosevich et al. 2003). Long-distance dispersal events can also produce scattered satellite populations (foci) that grow with time and accelerate the invasion rate (Minor & Gardner 2011). This process is not exclusive of invasive species. According to Higgins and Richardson (1999), the mismatch often found between observed plant migration rates (faster) and ecological spread models (slower) can be explained by the fact that rare long-distance dispersal events can lead to substantial increase in the spread rate.

Without control, the source population continues to grow, and so do the foci that can become further sources of populations themselves. In fact, these small patches tend to expand more rapidly and cover a greater area than does the progressing front of a large patch (Cousens & Mortimer 1995). For that reason, the preferred containment strategy would be to remove the local satellite populations as soon as possible before they reach considerable growth rates (Moody & Mack 1988). Mack and Lonsdale (2002) argued that the ideal eradication campaign would be to destroy all the individuals of a potentially invasive species immediately upon their arrival or, if this approach fails, to remove all the small foci quickly. Also, the more established a population is, the more difficult it is to restore the area successfully. According to Holmes et al. (2000), an additional reason to clear the invaded sites early on is that the intensity of restoration intervention required is proportional to the invasion duration.

Although it is not an infallible solution or a generally recommended line of action, the bibliography suggests that removing more small patches of IAP rather than few large ones may, in many cases, be beneficial for the efforts of fighting the invasion (Campbell 1993). This work aims to identify which strategy is best, based on land-cover changes and landscape connectivity criteria, to halt *Acacia dealbata* invasion in a Natura 2000 site in central Portugal.

Methods

Study Area

The study area corresponds approximately to the Special Area of Conservation "Serra da Lousã" (PTCON0060) in central Portugal. Lousã mountain's highest peaks range between 800 and 1,200 m and display some very steep slopes and narrow valleys. The rough orography and influences by Atlantic and Mediterranean climates, contribute to diversified vegetation and make the site relevant from the landscape standpoint. The 15,157 ha of the SAC are almost entirely covered by forest (12,008 ha) and shrublands (2,423 ha). The forest is mainly formed by *Pinus pinaster* (58.5%) and *Eucalyptus globulus* (16.6%). The invasion by alien species is one of the main concerns, especially when considered together with the threat of wildfires, from which some of these species (e.g. *Acacia dealbata*, *Hakea sericea*) can capitalize to increase their distribution rapidly. Considerable efforts have been made to preserve the site's biodiversity and prevent further invasion. Measures to minimize invasion or to restore invaded areas are common to different projects and plans that coincide in the area, from broad forestry plans to explicit projects such as the recently concluded "GANHA—sustainable management of *Acacia* spp: natural control and further methods to restore habitats in classified areas" (https://www.invasoras.pt/pt/gest%C3%A3o-sustent%C3%A1vel-de-plantas-invasoras [accessed 21 May 2021]). GANHA project aimed to control 20 ha of *A. dealbata* and some additional *Acacia melanoxylon* foci in riparian galleries (Fig. 1).

Acacia dealbata. Australian acacias include some of the most important plant invaders globally (Richardson & Rejmánek 2011). Their vast seed banks enable them to dominate when an opportunity unveils, whether due to natural or
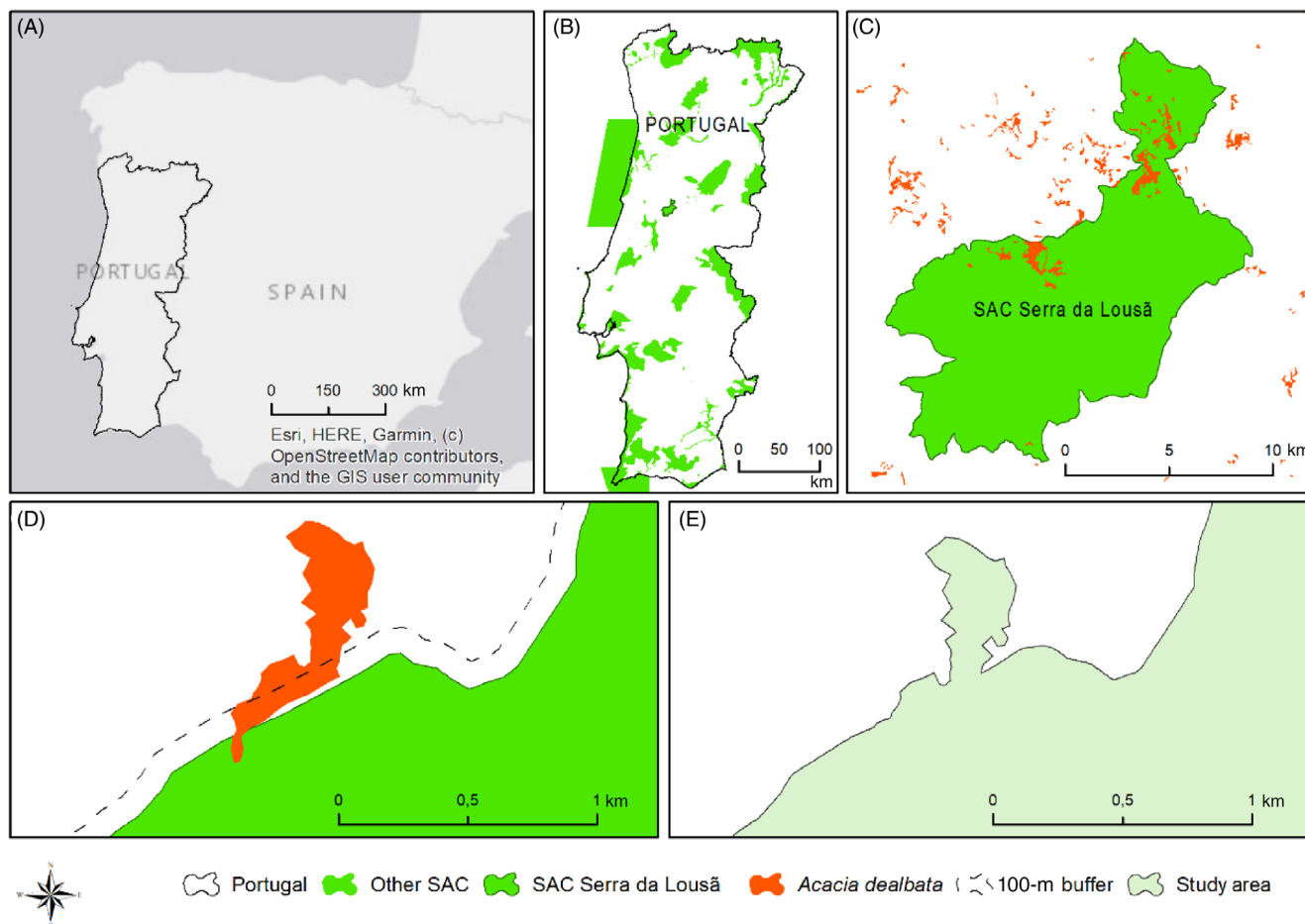
Figure 1. Study area. (A) Iberian Peninsula; (B) Special Areas of Conservation (SAC) in mainland Portugal; (C) SAC "Serra da Lousã" and *Acacia dealbata*; (D) detailed view of the 100-m buffer; (E) incorporation of the buffer and intersected *A. dealbata* patches in the study area.

anthropogenic disturbances (Lorenzo et al. 2010; Fuentes-Ramírez et al. 2011). The invasion usually results in landscape homogenization and associated loss of biodiversity and degradation of ecosystem services over time (le Maitre et al. 2011). The silver wattle *A. dealbata* was first introduced in Portugal in 1850 for ornamental reasons and soil stabilization. Nowadays, the species is present all over the country and known for its high invasive potential. The most extensive patches are primarily located in the north and center of Portugal. Still, it grows nation-wide along water stream banks and roadsides (Plantas Invasoras em Portugal 2020). Although the strips of IAPs in the stream banks and roadsides may cover small areas and therefore appear harmless or least concerning, they have elongated shapes and consequently high PAR. They can also act as a reservoir of propagules that can be liberated in disturbance events (Parendes & Jones 2000) and are usually important vectors of invasion into protected areas (Landres et al. 1998).

Regarding dispersal processes and patterns, the *A. dealbata* seed bank is mainly located under the tree canopy and its density declines steeply away from canopy (Passos et al. 2017). The seeds are mainly ant-dispersed which also limits their spreading distance (Gibson et al. 2011). Exceptions to these short-distance

dispersal processes are the casual dispersal by humans (seeds inadvertently transported in clothes, tools, machinery, etc.) and the abiotic dispersal by water. Overall, *A. dealbata* is more prone to advance as a front than to display frequent long-dispersal events. In this regard, Minor and Gardner (2011) found that species with a high probability of random long-distance dispersal are best managed by focusing on the largest patches, while species more prone to short-distance dispersion are best managed considering the landscape configuration of the patches.
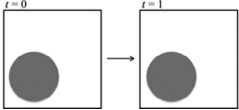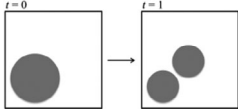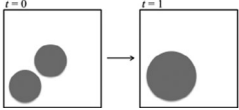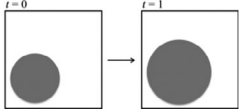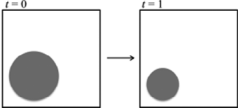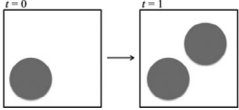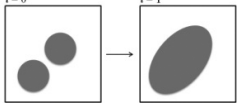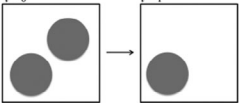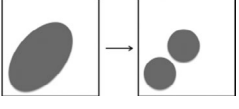
Base Maps, Preliminary Steps, and Scenarios

The 2018 official land-cover map of Portugal (Carta de Ocupação do Solo 2018) was used to extract the invasive species patches (Direção-Geral do Território 2019). Due to its detailed and heterogeneous land-cover classification scheme, quality, and reliability, this cartography is widely used to support the most relevant land planning and management-related policy procedures and scientific studies developed in mainland Portugal. The map has a minimum map unit of 1 ha and a minimum distance between lines of 20 m. The land-cover category

"Invasive Species Forest" includes more than one species. After exploring the area, it was noticeable that *A. dealbata* is by far the most abundant invasive species, and the same is corroborated by the Global Biodiversity Information Facility (https://www.gbif.pt/). Considering the minimum map unit of 1 ha, it is safe to assume that most—if not all—the patches classified as "Invasive Species Forest" represent *A. dealbata*. Thus we assume, in the context of this methodological demonstration, the land-use category "Invasive Species Forest" as *A. dealbata* forest patches.

Table 1. Landscape dynamic types (adapted from Machado et al. 2018). Variation in area and number of patches (NP) in a spatial extent between two moments may originate different types of dynamics.

ToD	ΔArea	ΔNP	Designation	Spatial pattern
A	=0	=0	No change	
B	=0	>0	Fragmentation per se	
C	=0	<0	Aggregation per se	
D	>0	=0	Gain	
E	<0	=0	Loss	
F	>0	>0	NP increment by gain	
G	>0	<0	Aggregation by gain (NP decrement by gain)	
H	<0	<0	NP decrement by loss	
I	<0	>0	Fragmentation by loss (NP increment by loss)	
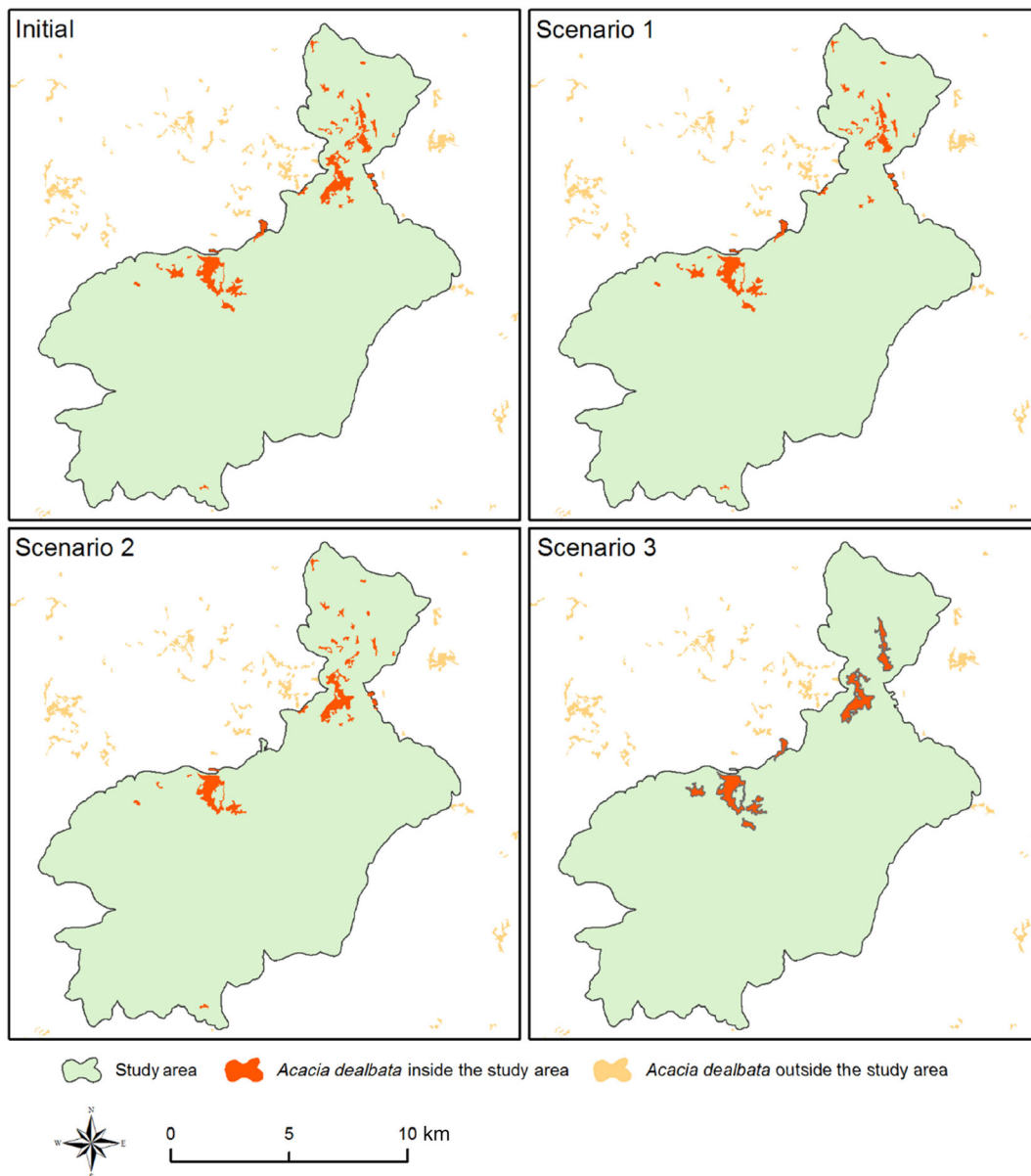
Figure 2. Initial and scenarios' maps. In all scenarios the area of *Acacia dealbata* removed is similar. In scenario 1 the largest patch is removed. In scenario 2, five patches with intermediary perimeter–area ratio values are removed. In scenario 3, the 25 patches with higher perimeter–area ratio are removed.

The study area was defined based on the official Natura 2000 Network map, namely the Special Areas of Conservation (SAC) shapefile. We extracted the SAC "Serra da Lousã" (PTCON0060), applied an outward 100-m buffer, and included the acacia patches intersected by the buffer. The decision to include the nearest patches outside the SAC is based on the premise that removing the invader within the SAC and leaving patches in the adjacent area would probably result in reinvasion in the short term (Landres et al. 1998). This way, we avoid over-looking the processes occurring in nearby territory that may be influencing the landscape patterns in the SAC (Pauchard et al. 2003).

Table 2. Area, edge length, and number of patches in the initial situation and in the three scenarios. The values in brackets are the differences between the scenario and the initial situation. Numbers in bold show the edge length and number of patches (NP) are more affected in scenario 3.

	Initial	*Scenario 1*	*Scenario 2*	*Scenario 3*
Area (ha)	429.76	323.14 (−106.62)	322.26 (−107.49)	330.82 (−98.94)
Edge length (m)	75,760	62,279 (−13,481)	57,700 (−18,060)	**45,848 (−29,912)**
NP	33	32 (−1)	28 (−5)	**8 (−25)**
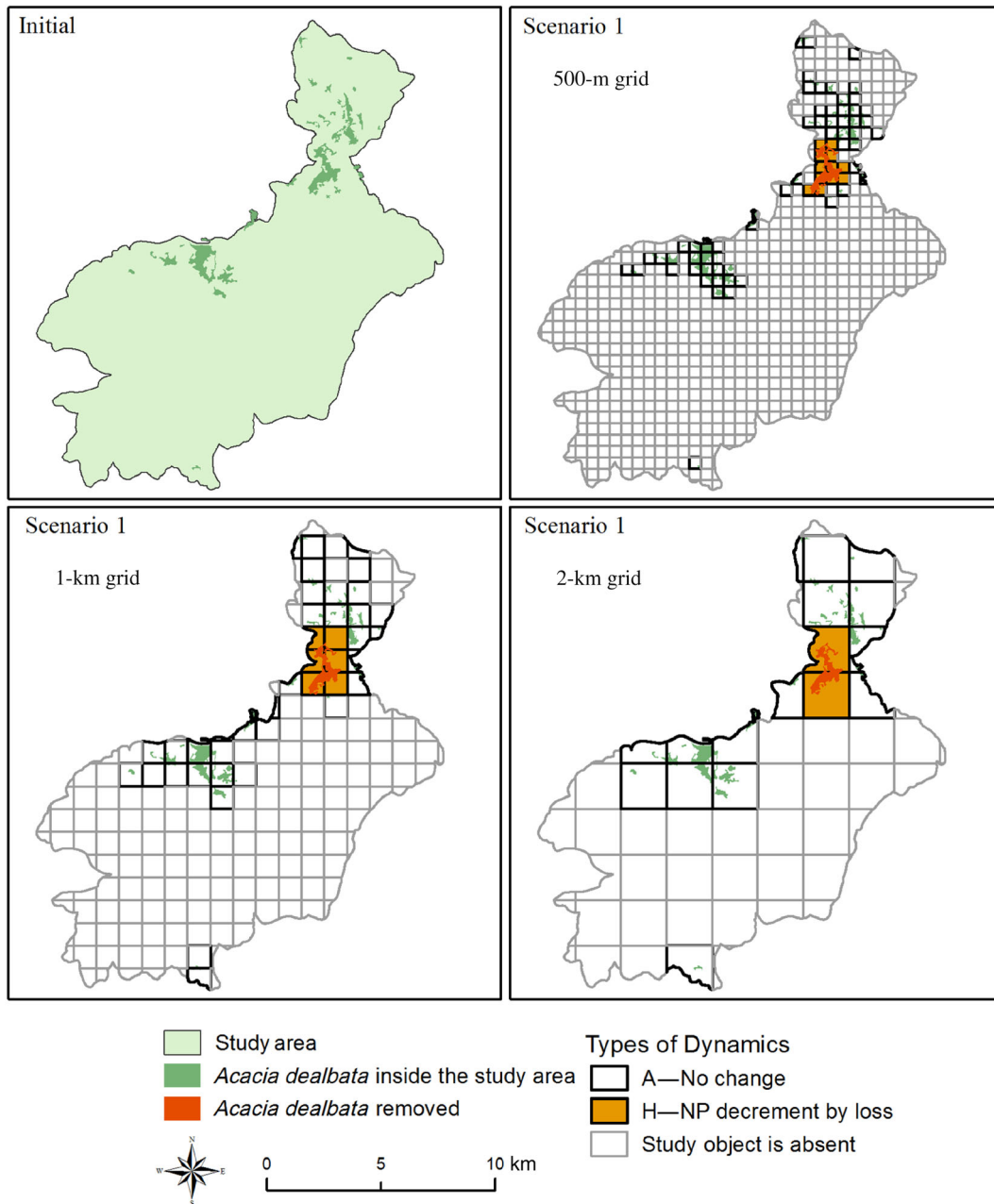
Figure 3. Types of dynamics for scenario 1 calculated using different grid sizes.

The first task was to calculate the PAR for every *Acacia* patch in the SAC. Then, three intervention scenarios were defined. The first scenario involves removing the largest *A. dealbata* patch (area = 106.61 ha). The second scenario involves removing intermediary PAR patches that total an area as close as possible to the area of the largest patch ($n = 5$; total area = 107.49 ha). The third scenario involves removing the highest PAR patches up to an area as close as possible to the area of the largest patch ($n = 25$; total area = 98.94 ha).

Landscape Dynamics Analysis

To know what type of dynamics the different interventions would produce, considering landscape composition and configuration, we ran the simulations using LDTtool in ArcGIS 10.7 (Machado et al. 2020) (Table 1). Because we did not know beforehand which spatial resolution would be appropriate, we calculated the resulting types of dynamic for each scenario using 500×500–m², $1,000 \times 1,000$–m², and $2,000 \times 2,000$–m² grids.
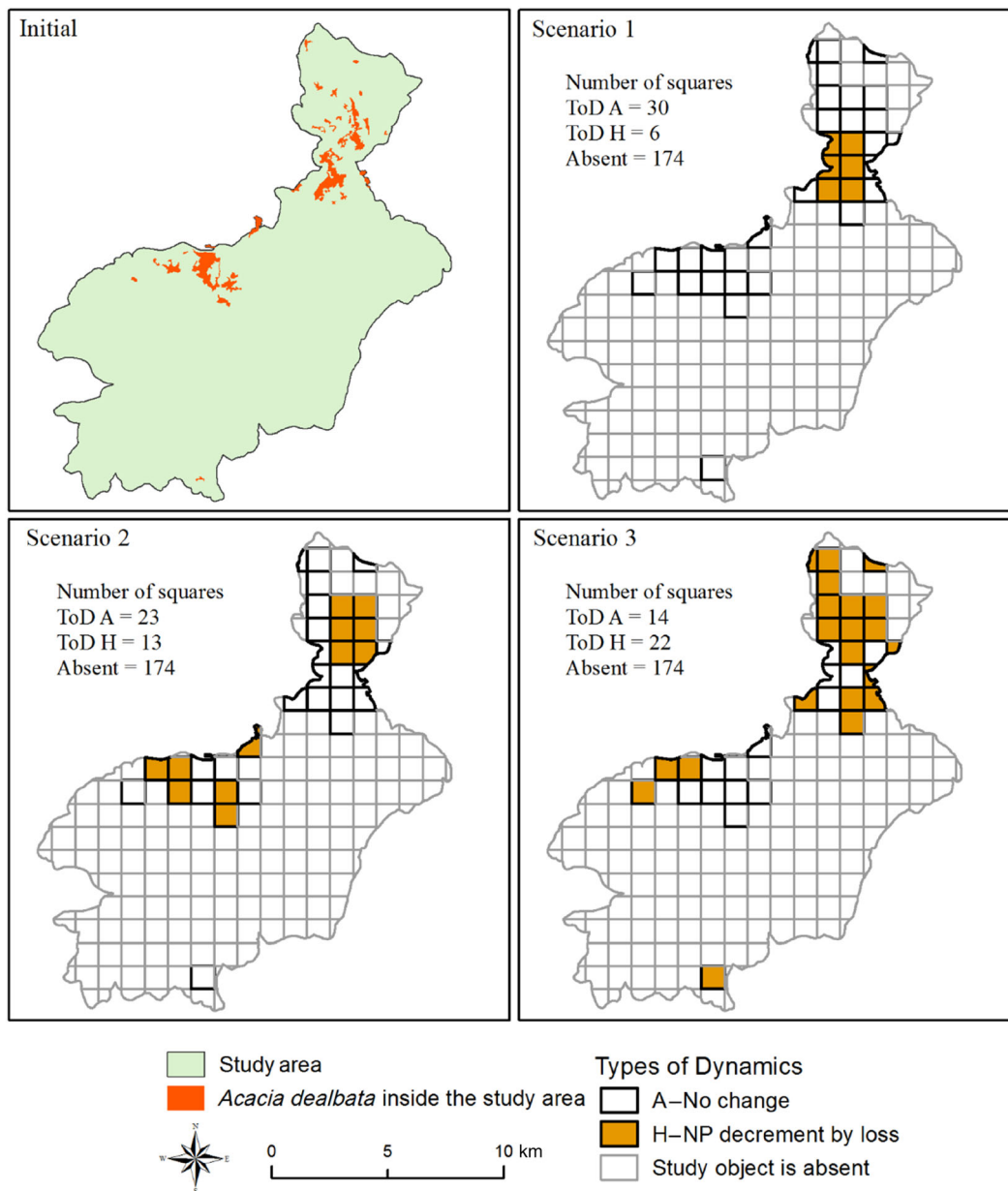
Figure 4. Types of dynamics produced by the different scenarios.

Connectivity Analysis

Once there is a set of patches identified for removal, several criteria can be used to rank them and help decide where to act first. A valuable aid would be to know beforehand how important each patch is for the species' connectivity in the study area. We assessed each patch's role to connectivity using the Conefor Sensinode software (Saura & Torné 2009) via the metric:

$$\mathrm{d}M\,(\%) = 100.\frac{M - M_{\text{after}}}{M}$$

where M is an overall connectivity metric when all patches are present in the landscape and M_{after} is the metric value after a determined patch is removed. Running the simulation for all *A. dealbata* patches, we obtain each patch contribution to the species connectivity in the study area. The higher the dM, the more important the patch is for the connectivity and the higher it should be ranked in the removal list. As overall connectivity metric (M), we used the numerator of the integral index of connectivity (IIC$_{\text{num}}$) (Pascual-Hortal & Saura 2006) given by:
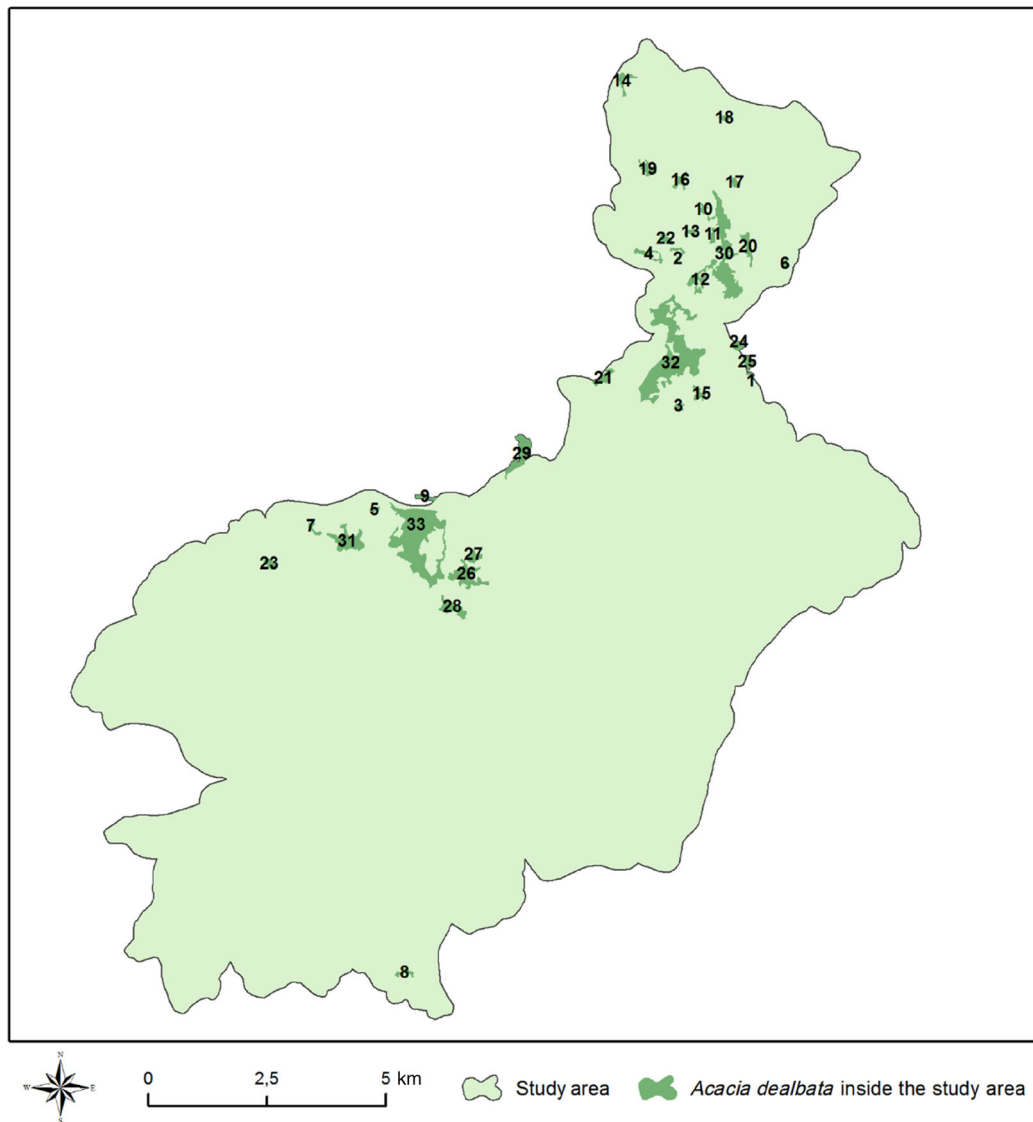
Figure 5. Ranking of *Acacia dealbata* patches based on their relative importance for the species' overall connectivity in the study area. Calculated using perimeter–area ratio as the key attribute. From 1—patch that contributes the most, to 33—patch that contributes the least.

$$\mathrm{IIC_{num}} = \sum_{i=1}^{n} \sum_{j=1}^{n} \frac{a_i . a_j}{1 + nl_{ij}}$$

where n is the total number of patches in the landscape, a_i and a_j are attributes of patches i and j, and nl_{ij} are the number of links between patches i and j. The threshold distance used was 100 m and PAR was pointed as the patch key attribute/characteristic.

Results

The analytical outcomes are (1) basic instrumental data intrinsic to each scenario (area, edge, and patches removed and remaining); (2) concrete results regarding landscape dynamics; and (3) complementary results based on the connectivity assessment; and (4) final list of patches to remove according to a combined analysis of the previous elements.

Scenarios

The existing situation and the resulting scenarios are spatially represented in Figure 2 and the associated values (and variations) of area, edge, and number of patches are present in Table 2. It is noticeable that for similar area amounts, the removal of many smaller patches instead of fewer larger ones causes the removal of a substantial extra edge length. For instance, removing the larger patch (scenario 1) subtracts 13,481 m of edge, while removing the smaller patches totaling a similar area (scenario 3) leads to a decrement of 29,912 m of edge.
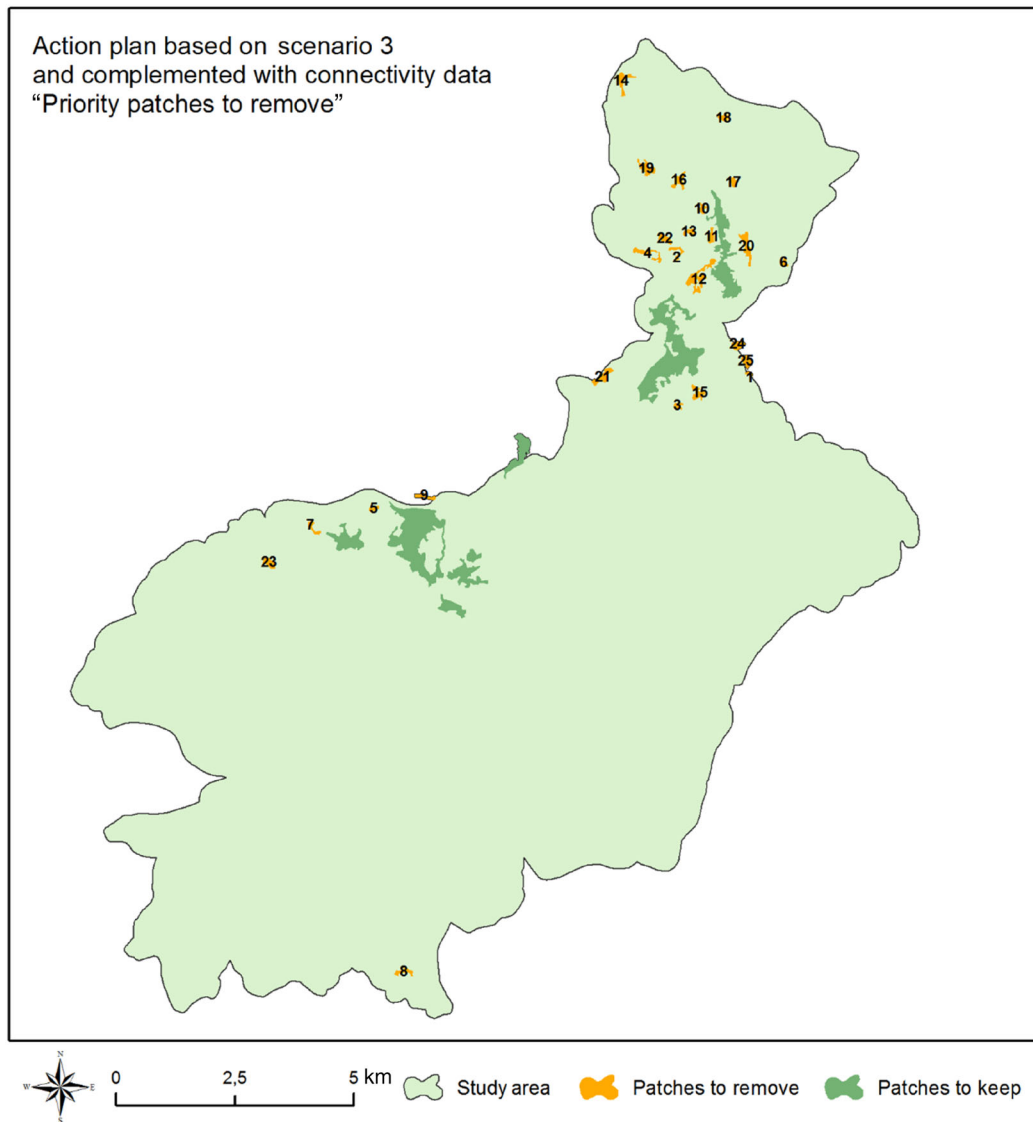
Figure 6. Ranking of *Acacia dealbata* patches to remove toward scenario 3 and considering connectivity aspects.

Landscape Dynamics Analysis

To ascertain which spatial resolution would fit the analysis better, all scenarios were run using 500-m, 1,000-m, and 2,000-m grids (Fig. 3). The smallest squares cover 2.5 ha each, and as the minimum map unit is 1 ha, the probability of a patch being included in a square is very low. These squares are only two and a half times bigger than the smallest patch; therefore, there will be a high number of patches that belong to more than one square, a situation that should be minimized to keep the analysis effective. The largest squares solve that problem but are so large that important changes could go unnoticed. This grid size is usable, but there is no reason to select it if it is possible to use a higher spatial resolution without compromising the analysis. Based on these reasons, we decided to analyze with the intermediary 1,000-m grid.

Looking at the results obtained using the 1,000-m grid, in scenario 1 there is Type of Dynamic H in few contiguous squares ($n = 6$), in scenario 2 there are more H squares ($n = 13$) and more spread across the study area, and finally, scenario 3 displays even a higher number of H squares ($n = 22$) (Fig. 4). Overall, for a similar area of intervention (approximately 106.62 ha), the elimination of a large patch constitutes a more localized intervention. In contrast, the elimination of multiple smaller patches means a more spatially widespread intervention.

Connectivity Analysis

The connectivity analysis reinforces the spatial aspects and provides information about patch importance for the species

connectivity. The output is a map where the *A. dealbata* patches are ranked according to how much they contribute to the species overall connectivity in the study area (Fig. 5).

Combined Analysis

The landscape dynamics analysis showed that scenario 3 produces more squares with ToD H—"NP decrement by loss" by removing many smaller patches than other scenarios where fewer but larger patches would be eliminated. The connectivity analysis showed that the patches have contributed differently, due to PAR and location, to the global species connectivity and may be ranked according to this criterion.

Combining both approaches, we propose a strategy to aim for scenario 3, which involves removing the higher PAR patches (mostly smaller patches), beginning with those with higher contribution to connectivity, as depicted in Figure 6.

Discussion

Restoring areas invaded by IAP is challenging and every measure to increase the success probability should be deployed. We highlight the importance of decisions that need to be made before the fieldwork starts: where to act (first) and why? In an IAP intervention program it is important to remove the more relevant patches early due to the difficulty to correctly estimate the required investment needed to succeed (Panetta 2009). A patch ranking list can help managers decide which patches to eliminate first, mainly because the resources may not be enough to eliminate all the patches or if the task is not expected to be completed in a single period. Ultimately, it is up to the manager to decide what is feasible according to the available resources (time included). From the operational point of view, it might be easier to remove one large patch, but the opposite strategy of removing the smaller patches may be more effective to achieve the end goal. In this case study, scenario 2 illustrates how an intermediate compromise may be a solution. Removing several patches instead of just one poses logistical challenges (dispersed team, need for additional transportation, time spent cruising from one workplace to another, etc.) that must be taken into consideration. Thus, it is crucial to have a decision-support tool or procedure that provides insights of how far the efforts should go. For instance, choosing scenario 2 instead of scenario 1 would remove additional 4,689 m of edge length by subtracting four more patches. Implementing scenario 3 instead of scenario 2 would require the removal of 20 more patches to reduce the edge by an additional 11,852 m. It is up to the manager to establish the cost–benefit threshold for the contexts at hand.

In general terms, the control of invasive species is intended to reverse the associated impact by recovering the structure and composition of natural communities and reducing the area covered and preventing their expansion (Zalba & Ziller 2007). In our case, preventing the expansion is more efficiently achieved by eliminating the higher PAR patches due to their potential to boost the invasion rate.

More foci mean greater opportunity for more spatial connectivity that, in time, may lead to an exponential increase in spread

rates (Higgins & Richardson 1999; Doren et al. 2009; Hernández et al. 2014). Therefore, one advantage of removing satellite patches first is that it prevents them from merging with other satellite patches or from being absorbed by the parental patch in the future (Pauchard et al. 2003). That situation, represented by ToD G—Aggregation by gain, provokes landscape homogenization and therefore should be prevented. Conversely, if invasion rate is more worrisome than homogenization, removing a patch that belongs to a cluster of patches, even if it highly contributes to connectivity, may not be a priority. In such case, the (configuration/geometric) expansion potential would be naturally limited due to the proximity to other IAP patches while isolated IAP patches could represent a more significant chance to quickly and vastly occupy adjacent territory. Such a situation is present in the northern part of the study area where some patches are clustered and removing some of them can make sense to affect the IAP connectivity but do not necessarily represent the major impact in terms of halting the potential spreading.

Mack and Lonsdale (2002) alerted that ignoring small foci of IAP while focusing on major infestations provides time for the once-inconspicuous satellite populations to flourish. A clear example is *Schinus terebinthifolius*, which was introduced as an ornamental to South Florida but did not explode across the landscape until decades later (Ewel 1986). Many widely separated foci can be more difficult to eradicate than a single larger infestation but detection and eradication of all nascent foci may be more important than attacking large centers of the invasion (Mack & Lonsdale 2002). Konlechner et al. (2015) went even further, stating that managers should prioritize preventing its spread into uninvaded areas over its removal from invaded sites once an invasive species has been established. A practical example of how satellite populations can be a more significant threat than the parental patch was provided by Ghersa et al. (2000). After studying Johnsongrass *Sorghum halepense*, they observed that small foci uniformly distributed over a previously vacant area occupied that area more quickly than did the advancing front of an adjacent source population.

Our proposal for dealing with *Acacia dealbata* in the SAC "Serra da Lousã" follows the same line of thought of removing the higher PAR, and thus mostly the smaller patches. Among them, those potentially boosting ToD G—"Aggregation by gain" should be eliminated first. Once the small foci are removed and the probability of rapid expansion is lowered, one can focus on the large patches. In many cases, eradication is unlikely, but area reduction and prevention of further spreading are achievable goals. In this particular case of invasion by *A. dealbata*, the edge must be controlled using adequate techniques according to the plant's size: herbicide spraying of saplings and herbicide application after cutting of adult plants (le Maitre et al. 2011; Souza-Alonso et al. 2013). Preferably, saplings should be hand-pulled and larger trees debarked (i.e. removes bark and cambium layers to the ground cutting the nutrients flow and killing the roots) (personal observation).

The adjacent land-covers have different permeability to the invasion, and that must be taken into consideration. Focusing on forests because that is the main cover of the study area, it was suggested that invasions are hindered by the edge response

of the adjacent patches, especially those with a dense wall of bordering vegetation that reduces interior light levels and wind speeds (Brothers & Spingarn 1992). Cadenasso and Pickett (2001) clearly stated that a way to fight invasion expansion is to keep forest edges intact to function as a barrier to the flux of seeds while actively removing the established IAP on the edge. An associated question is: how far in goes the edge effect? It depends on several factors such as species, age, and density, among others, but it is always influenced by the patch size and mainly by its shape (Laurance & Yensen 1991). For that reason, landscape configuration should be considered when selecting restoration measures (Reis et al. 2020). In our case study, the IAP patches were almost entirely surrounded by forest, mostly pine, and only one was completely surrounded by shrublands and thus supposedly facing less resistance to expansion. We had no further information about the stand age, density, edge composition, or other variables that could be used as additional criterion to enrich the ranking process.

It is essential to mention that although this work theorizes, points strategic lines, and suggests concrete actions; it is not an actual plan for the SAC. The main reason why this cannot be seen as a plan is because that would require more accurate and updated data. A proper inventory of the populations is vital to avoid missing isolated patches or individuals that can then invalidate the possibility of success (Hélia et al. 2019).

We used the official land cover map of mainland Portugal, COS 2018 (Direção-Geral do Território 2019), which has a minimum map unit of 1 ha and does not represent (highly relevant) smaller patches. According to Martins et al. (2016), who mapped *A. dealbata* in a nearby location using remote sensing techniques, the species is spread in patches smaller than 0.5 ha. Consequently, the invaded area is expected to be much higher than reported in the official maps and statistics. Recently, Ferreira et al. (2021) also compiled a map using several sources, and their patch size evaluation based on photointerpretation of orthophotomaps from Portugal included a class of patches smaller than 0.1 ha. Another aspect to consider is that despite the advances in remote sensing technology, ground surveys can still play a relevant role in the detection of new source and satellite populations (Radosevich et al. 2003). Therefore, thorough fieldwork would be fundamental to complement remote sensing-based maps and ultimately obtain a more reliable representation of the actual situation in the terrain.

We suggest this approach for this species in this context and thus the analytical procedure here presented should not be seen as a "one-size-fits-all" approach. The location of management actions may be selected according to several criteria. We used a spatially explicit strategy focused on the satellite populations and incorporated connectivity information to enrich the analysis and support better decision-making. However, different contexts and species demand different approaches. An alternative option would be to focus the control on large populations because they produce the most significant number of dispersal propagules to the next generation (more suitable for long-distance dispersal species) (e.g. Shmida & Ellner 1984). Another approach related to connectivity would be integrating

dispersal biology into the management strategies, such as limiting access to vector pathways (e.g. roadsides) and focusing on specific vectors, as discussed by Davies and Sheley (2007). Weighing the pros and cons and considering the resources and constraints, the manager should be able to make an informed decision and adopt an appropriate strategy.

This analytical method can be applied to large extensions of terrain but that does not mean that is the best approach. The procedure will always calculate the ToD and connectivity, regardless of the study area size but since the resources are usually scarce, the area can be divided in sectors based on other, more operational, criteria. A national scale ranking of patches is not necessarily more useful than the same type of information on a more workable scale, say regional or local. As Krug et al. (2010) stated: "The financial resources available determine the extent of the area which can be cleared, while the prioritization identifies the location of the areas to be cleared." For that reason, there is no need to prioritize an area much larger than the area that can be covered with the actual resources (e.g. estimate to clear 100 ha and conduct an analysis to prioritize 10,000 ha).

A useful case we can anticipate for this method is the screening of large areas in early phases to support decision-making afterward. For example, if the graph-based connectivity analysis identifies large *A. dealbata* components or "connected regions" (groups of patches isolated from the other patches; Pascual-Hortal & Saura 2006) in the landscape it could make sense to focus efforts on a single component, prioritize its patches according to the criteria found relevant and act to prevent ToD G—"Aggregation by gain."

Although developed based on a case study in Portugal, provided that the principles and main premises apply, this method is viable for use in different geographic contexts for this species (and others with similar dispersal behavior). Since it is based on spatial metrics (NP, area, and PAR) and their relations, the rationale is transferable to other similar contexts. However, exceptions may apply (e.g. due to differences in environmental conditions) and managers should be aware that regardless of how useful a method may seem it is important to adjust prioritization strategies to the particulars of each region (Roura-Pascual et al. 2010). In summary, our method for supporting the control planning of *A. dealbata* constitutes a strategy based on landscape dynamics fine-tuned with complementary connectivity information so that we end up with a clear, ordered list of patches to remove.

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LITERATURE CITED

Bellard C, Cassey P, Blackburn TM (2016) Alien species as a driver of recent extinctions. Biology Letters 12: 20150623, https://royalsocietypublishing.org/doi/pdf/10.1098/rsbl.2015.0623

Brothers T, Spingarn A (1992) Forest framentation and alien plant invasion of Central Indiana old-growth forests. Conservation Biology 6:91–100

Butchart SHM, Walpole Matt, Collen Ben, van Strien Arco, Scharlemann Jörn PW, Almond Rosamunde EA, et al. (2010) Global biodiversity: indicators of recent declines. Science 328:1164–1168

Cadenasso ML, Pickett STA (2001) Effect of edge structure on the flux of species into forest interiors. Conservation Biology 15:91–97

Callaway RM, Aschehoug ET (2000) Invasive plants versus their new and old neighbors: a mechanism for exotic invasion. Science 290:521–523

Campbell P (1993) Wattle control. Handbook no. 3. Plant Protection Research Institute, Pretoria, South Africa

Cheney C, Esler KJ, Foxcroft LC, van Wilgen NJ (2019) Scenarios for the management of invasive Acacia species in a protected area: Implications of clearing efficacy. Journal of Environmental Management 238:274–282.

Corbin JD, D'Antonio CM (2012) Gone but not forgotten? Invasive plants' legacies on community and ecosystem properties. Invasive Plant Science and Management 5:117–124

Cordell S, Ostertag R, Michaud J, Warman L (2016) Quandaries of a decade-long restoration experiment trying to reduce invasive species: beat them, join them, give up, or start over? Restoration Ecology 24:139–144

Cousens R, Mortimer M (1995) Dynamics of weed populations. Cambridge University Press, Cambridge, United Kingdom

D'Antonio C, Meyerson LA (2002) Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restoration Ecology 10:703–713

Davies KW, Sheley RL (2007) A conceptual framework for preventing the spatial dispersal of invasive plants. Weed Science 55:178–184

Diagne C, Leroy B, Gozlan RE, Vaissière AC, Assailly C, Nuninger L, Roiz D, Jourdain F, Jarić I, Courchamp F (2020) InvaCost, a public database of the economic costs of biological invasions worldwide. Scientific Data 7: 1–12

Diagne C, Leroy B, Vaissière AC, Gozlan RE, Roiz D, Jarić I, Salles JM, Bradshaw CJA, Courchamp F (2021) High and rising economic costs of biological invasions worldwide. Nature 592:571–576

Direção-Geral do Território (2019) Especificações técnicas da Carta de uso e ocupação do solo (COS) de Portugal Continental para 2018. Relatório Técnico.

Doren RF, Richards JH, Volin JC (2009) A conceptual ecological model to facilitate understanding the role of invasive species in large-scale ecosystem restoration. Ecological Indicators 9:150–160

Ewel JJ (1986) Invasibility: lessons from South Florida. Pages 214–230. In: Mooney HA, Drake JA (eds) Ecology of biological invasions of North America and Hawaii. Springer-Verlag, New York

Ferreira V, Figueiredo A, Graça MAS, Marchante E, Pereira A (2021) Invasion of temperate deciduous broadleaf forests by N-fixing tree species—consequences for stream ecosystems. Biological Reviews 96:877–902

Fuentes-Ramírez A, Pauchard A, Cavieres LA, García RA (2011) Survival and growth of Acacia dealbata vs. native trees across an invasion front in south-central Chile. Forest Ecology and Management 261:1003–1009

Gaertner M, Nottebrock H, Fourie H, Privett SDJ, Richardson DM (2012) Plant invasions, restoration, and economics: perspectives from South African fynbos. Perspectives in Plant Ecology, Evolution and Systematics 14:341–353

Ghersa CM, Benech-Arnold RL, Satorre EH, Martínez-Ghersa MA (2000) Advances in weed management strategies. Field Crops Research 67:95–104

Gibson MR, Richardson DM, Marchante E, Marchante H, Rodger JG, Stone GN, et al. (2011) Reproductive biology of Australian acacias: important mediator of invasiveness? Diversity and Distributions 17:911–933

Gioria M, Pyšek P (2015) The legacy of plant invasions: changes in the soil seed bank of invaded plant communities. Bioscience 66:40–53

Gratton C, Denno RF (2005) Restoration of arthropod assemblages in a Spartina salt marsh following removal of the invasive plant Phragmites australis. Restoration Ecology 13:358–372

Harms RS, Hiebert RD (2006) Vegetation response following invasive tamarisk (Tamarix spp.) removal and implications for riparian restoration. Restoration Ecology 14:461–472

Hélia Marchante, Palhas Jael, Núñez Francisco Alejandro López, Marchante Elizabete (2019) Invasive species impacts and management. Pages 1–12. In: Leal Filho W, Azul A, Brandli L, Özuyar P, Wall T (eds) Life on land. Encyclopedia of the UN sustainable development goals. Springer, Cham, Switzerland

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. Forest Ecology and Management 329:206–213

Higgins SI, Richardson DM (1999) Predicting plant migration rates in a changing world: the role of long-distance dispersal. American Naturalist 153:464–475

Higgins SI, Richardson DM, Cowling RM (2000) Using a dynamic landscape model for planning the management of alien plant invasions. Ecological Applications 10:1833–1848

Holmes PM, Richardson DM, Wilgen BW, Gelderblom C (2000) Recovery of South African fynbos vegetation following alien woody plant clearing and fire: implications for restoration. Austral Ecology 25:631–639

Holmes PM, Esler KJ, van Wilgen BW, Richardson DM (2020) Ecological restoration of ecosystems degraded by invasive alien plants in South African fynbos: is spontaneous succession a viable strategy? Transactions of the Royal Society of South Africa 75:111–139

Hulme PE (2006) Beyond control: wider implications for the management of biological invasions. Journal of Applied Ecology 43:835–847

IPBES (2019) In: Brondizio ES, Settele J, Díaz S, Ngo HT (eds) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES Secretariat, Bonn, Germany

Jeschke JM, Bacher S, Blackburn TM, Dick JTA, Essl F, Evans T, et al. (2014) Defining the impact of non-native species. Conservation Biology 28: 1188–1194

Jones BA (2017) Invasive species impacts on human well-being using the life satisfaction index. Ecological Economics 134:250–257

Konlechner TM, Hilton MJ, Lord JM (2015) Plant community response following the removal of the invasive Lupinus arboreus in a coastal dune system. Restoration Ecology 23:607–614

Krug RM, Roura-Pascual N, Richardson DM (2010) Clearing of invasive alien plants under different budget scenarios: using a simulation model to test efficiency. Biological Invasions 12:4099–4112

Kumschick S, Gaertner M, Vilà M, Essl F, Jeschke JM, Pyšek P, et al. (2015) Ecological impacts of alien species: quantification, scope, caveats, and recommendations. BioScience 65:55–63

Landres PB, Knight R, Pickett S, Cadenasso ML (1998) Ecological effects of administrative boundaries. Pages 39–64. In: Knight RL, Landres PB (eds) Stewardship across boundaries. Island Press, Washington D.C.

Laurance WF, Yensen E (1991) Predicting the impacts of edge effects in fragmented habitats. Biological Conservation 55:77–92

Levine JM, Vilà M, Antonio CMD, Dukes JS, Grigulis K, Lavorel S (2003) Mechanisms underlying the impacts of exotic plant invasions. Proceedings of the Royal Society B: Biological Sciences 270:775–781

Lorenzo P, González L, Reigosa MJ (2010) The genus Acacia as invader: the characteristic case of Acacia dealbata Link in Europe. Annals of Forest Science 67:101–101

Machado R, Godinho S, Pirnat J, Neves N, Santos P (2018) Assessment of landscape composition and configuration via spatial metrics combination: conceptual framework proposal and method improvement. Landscape Research 43:652–664

Machado R, Bayot R, Godinho S, Pirnat J, Santos P, de Sousa-Neves N (2020) LDTtool: a toolbox to assess landscape dynamics. Environmental Modelling and Software 133:104847

Mack RN, Lonsdale WM (2002) Eradicating invasive plants: hard-won lessons for islands. Pages. 164–172. In: Veitch CR, Clout MN (eds) Turning the tide: the eradication of invasive species

le Maitre DC, Gaertner M, Marchante E, Ens EJ, Holmes PM, Pauchard A, et al. (2011) Impacts of invasive Australian acacias: implications for management and restoration. Diversity and Distributions 17:1015–1029

Marchante E, Kjøller A, Struwe S, Freitas H (2008) Invasive *Acacia longifolia* induce changes in the microbial catabolic diversity of sand dunes. Soil Biology and Biochemistry 40:2563–2568

Elizabete Marchante, Marchante Hélia, Freitas Helena, Kjøller Annelise, Struwe Sten (2019) Decomposition of an N-fixing invasive plant compared with a native species: consequences for ecosystem. Applied Soil Ecology 138:19–31

Martins F, Alegria C, Gil A (2016) Mapping invasive alien *Acacia dealbata* Link using ASTER multispectral imagery: a case study in central-eastern of Portugal. Forest Systems 25(3), e078

Minor ES, Gardner RH (2011) Landscape connectivity and seed dispersal characteristics inform the best management strategy for exotic plants. Ecological Applications 21:739–749

Moody ME, Mack RN (1988) Controlling the spread of plant invasions: the importance of nascent foci. Journal of Applied Ecology 25:1009–1021

Duarte LN, Pinto Gomes C, Marchante H, Marchante E (2020) Integrating knowledge of ecological succession into invasive alien plant management: a case study from Portugal. Applied Vegetation Science 23:328–339

Panetta FD (2007) Evaluation of weed eradication programs: containment and extirpation. Diversity and Distributions 13:33–41

Panetta FD (2009) Weed eradication—an economic perspective. Invasive Plant Science and Management 2:360–368

Parendes LA, Jones JA (2000) Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conservation Biology 14:64–75

Parkes J, Panetta F (2009) Eradication of invasive species: progress and emerging issues in the 21st century. Pages 47–60. In: Clout M, Williams Peter A (eds) Invasive species management. A handbook of principles and techniques. Oxford Unviersity Press, Oxford, United Kingdom

Pascual-Hortal L, Saura S (2006) Comparison and development of new graph-based landscape connectivity indices: towards the priorization of habitat patches and corridors for conservation. Landscape Ecology 21:959–967

Passos I, Marchante H, Pinho R, Marchante E (2017) What we don't seed: the role of long-lived seed banks as hidden legacies of invasive plants. Plant Ecology 218:1313–1324

Pauchard A, Alaback PB, Edlund EG (2003) Plant invasions in protected areas at multiple scales: *Linaria vulgaris* (Scrophulariaceae) in the West Yellowstone area. Western North American Naturalist 63:416–428

Plantas Invasoras em Portugal (2020) *Acacia dealbata*. https://invasoras.pt/pt/planta-invasora/acacia-dealbata (accessed 21 May 2021)

Pyšek P, Hulme PE, Simberloff D, Bacher S, Blackburn TM, Carlton JT, et al. (2020) Scientists' warning on invasive alien species. Biological Reviews 95:1511–1534

Radosevich SR, Stubbs MM, Ghersa CM (2003) Plant invasions—process and patterns. Weed Science 51:254–259

Reis BP, Kövendi-Jakó A, Szitár K, Török K, Halassy M (2021) Long-term effect of mowing on the restoration of Pannonian sand grassland to replace invasive black locust plantation. Restoration Ecology 29:e13152

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

Roura-Pascual N, Richardson DM, Krug RM, Brown A, Chapman RA, Forsyth GG, et al. (2009) Ecology and management of alien plant invasions in South African fynbos: accommodating key complexities in objective decision making. Biological Conservation 142:1595–1604

Roura-Pascual N, Krug RM, Richardson DM, Hui C (2010) Spatially-explicit sensitivity analysis for conservation management: exploring the influence of decisions in invasive alien plant management. Diversity and Distributions 16:426–438

Saura S, Torné J (2009) Conefor Sensinode 2.2: a software package for quantifying the importance of habitat patches for landscape connectivity. Environmental Modelling and Software 24:135–139

Schindler S, Staska B, Adam M, Rabitsch W, Essl F (2015) Alien species and public health impacts in Europe: a literature review. NeoBiota 27:1–23

Shmida A, Ellner S (1984) Coexistence of plant species with similar niches. Vegetatio 58:29–55

Simberloff D, Martin JL, Genovesi P, Maris V, Wardle DA, Aronson J, et al. (2013) Impacts of biological invasions: what's what and the way forward. Trends in Ecology & Evolution 28:58–66

Souza-Alonso P, Lorenzo P, Rubido-Bará M, González L (2013) Effectiveness of management strategies in *Acacia dealbata* Link invasion, native vegetation and soil microbial community responses. Forest Ecology and Management 304:464–472

Turner MG, Hargrove William W, Gardner Robert H, Romme William H (1994) Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. Journal of Vegetation Science 5:731–742

Van Kleunen M, Weber E, Fischer M (2010) A meta-analysis of trait differences between invasive and non-invasive plant species. Ecology Letters 13:235–245

Vilà M, Espinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, 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

Vosse S, Esler KJ, Richardson DM, Holmes PM (2008) Can riparian seed banks initiate restoration after alien plant invasion? Evidence from the Western Cape, South Africa. South African Journal of Botany 74:432–444

van Wilgen BW, Forsyth Greg G, Le Maitre David C, Wannenburgh Andrew, Kotzé Johann DF, van den Berg Elna, Henderson Lesley (2012) An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. Biological Conservation 148:28–38

Zalba SM & Ziller S (2007) Adaptive management of alien invasive species: putting the theory into practice. Natureza & Conservação 5:86–92

Zedler JB, Callaway JC (2000) Evaluating the progress of engineered tidal wetlands. Ecological Engineering 15:211–225

RESEARCH ARTICLE

Using graph theory to analyse and assess changes in Mediterranean woodland connectivity

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Abstract

Context The Portuguese *montado* is an agro-silvopastoral system, similar to the Spanish *dehesa*, known for its cultural, economic and ecological value. Despite its importance, contrasting processes such as land abandonment and land use intensification, together with several other factors, have been responsible for *montado* degradation in the last decades. Biodiversitywise, assuring high levels of connectivity is vital for many species that, in turn, contribute to the natural processes on which a healthy and sustainable *montado* relies.

Objectives To study the *montado* connectivity in the recent decades and infer what the changes represent to the short and medium dispersal species regarding habitat availability.

Methods The study was conducted in an area delimited by biogeographic boundaries in Southern Portugal where *montado* is abundant. We used a graph theory based approach and *montado* maps of 1984, 1999 and 2014 derived from remote sensing.

Results The results show a loss of *montado* associated to increasing fragmentation over time. This led to a global connectivity decrement likely to have negative implications for *montado* species. The most affected species are those more dependent on habitat characteristics, such as forest specialist birds, and those with low mobility that have lost great amounts of habitat not only due to *montado* loss but also due to the increasing fragmentation that makes suitable patches unreachable.

Conclusions Given the *montado* environmental relevance, measures should be taken in order to stop its loss and preserve the core areas that have guaranteed the connectivity over time.

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Introduction

Landscape connectivity, defined by Taylor et al.
(1993) as the degree to which the landscape facilitates
or impedes the movement among resource patches, is
vital for ecosystems stability (With et al. 1997;
Collinge 1998). Connectivity has direct influence in
animal movements and population persistence (Johnson et al. 1992) and thus, is a major issue for wildlife
survival and biodiversity conservation (Fahrig and
Merriam 1985). Concerning plant populations and
communities, different responses can be found to
landscape/habitat connectivity depending, on their
character (positive or negative), of the plant functional
traits, life stages, matrix permeability and/or disturbance regime (Ibáñez et al. 2014). Although some
species may benefit from patch isolation as a result of
decreasing predation frequency or herbivory intensity
(e.g. Farwig et al. 2009), the negative effects are more
prevalent. Patch isolation may negatively affect
genetic flow (e.g. Steffan-Dewenter and Tscharntke
1999; Seltmann et al. 2009) or seed dispersal (e.g.
Kiviniemi 2008; Herrera and García 2010); and
barochoric (gravity-dispersed plants) and self-incompatible species are particularly susceptible to fragmentation (McEuen and Curran 2004; Aguilar et al.
2006; Lopes and Buzato 2007). However, zoochoric
plant species may also present limitations as a result of
connectivity loss if they are dispersed over short
distances (French et al. 2011) or if the landscape
matrix is not sufficiently permeable to allow animal
movements (Eycott et al. 2012; Astudillo et al. 2019).

Connectivity is central to the long-term persistence
of Iberian agro-silvopastoral landscapes dominated by
cork and holm oaks (e.g. Puerta-Piñero et al. 2012), in
the same way these land use systems (hereafter
montados or *dehesas*, designation used in Portugal
and Spain, respectively) are critical to achieve
sustainable territorial development in biophysically
constrained regions due to the multiple products,
values and services they provide (Martín Vicente and
Fernández Alés 2006; Surová et al. 2018). The low
intensity and balanced management associated with
the different vegetation layers promotes an heterogeneous landscape harboring biodiversity of high conservation value (Bugalho et al. 2011; Godinho et al.
2011), in accordance to the rule

"The low intensity greater the habitat variety, the
greater the species diversity" (Rosenweig 1995). For
instance, more than 95% of the terrestrial mammal
species of continental Portugal can be found, occasionally or frequently, in *montados* (Pinto-Correia
et al. 2013).

In this context, *montados* and *dehesas* are recognised as High Nature Value Farming systems, according to the European Environmental Agency
classification (Paracchini et al. 2008; Pinto-correia
et al. 2018). Despite these evidences, the sharp decline
of these unique open oak woodlands persists and their
long-term sustainability is threatened due to system
simplification resulting both from land use intensification in the more productive areas and land abandonment of the marginal ones (Pinto-Correia and
Mascarenhas 1999; Godinho et al. 2016c).

Montado fragmentation and lack of tree regeneration
can result directly from the use of heavy
machinery or due to grazing intensity above the
carrying capacity of the available resources (Dinis
et al. 2015; Almeida et al. 2016; Arosa et al. 2017;
Pinto-Correia et al. 2018). The cumulative and interactive effects of ecological and anthropic disturbance
regimes over a long time period have affected
significantly the regenerative capacity of Mediterranean ecosystems (Blondel 2006) and, therefore,
multiple recruitment limitations are also evident in
less intensively used areas covered by these evergreen
oak systems (Acácio et al. 2007).

Several authors argued that poor seed dispersal, low
seedling survival due to predation or long-term
drought, and scarcity of viable seeds are among the
constraints to tree recruitment (e.g. (Leiva and
Fernández-Alés 2003; Gómez-Aparicio et al. 2004;
Pulido and Díaz 2005). The dominant overstory
species (*Quercus rotundifolia*—or *Q. ilex*—, and
Q. suber) are mast-seeding evergreen trees (Pérez-
Ramos et al. 2010) and their dispersion occurs through
barochory and synzoochory mainly by the European
jay (*Garrulus glandarius*) and the wood mice (*Apodemus sylvaticus*) (Gómez 2003; Muñoz and Bonal
2007). Puerta-Piñero et al. (2012) stress that patch
connectivity has opposing indirect effects on holm
oaks recruitment and survival, by increasing the

activity of acorn dispersers like jays, but also of seed predators such as wild boars, which also have negative effects on the abundance of acorn-dispersing rodents (Muñoz et al. 2009). However, low patch connectivity seems to be more critical for acorn dispersion than for predation since the negative effects of wild boars on the abundance of holm oaks seedlings and saplings can be seen throughout the landscape and thus, are less dependent of patch connectivity (Puerta-Piñero 2010).

Management type and intensity can, in turn, mediate the effect of ecological disturbances on *montado* (Acácio et al. 2017), both abiotic such as drought (Camilo-Alves et al. 2017) and fires (Guiomar et al. 2015), and biotic such as plant pathogens (Camilo-Alves et al. 2013) or insect pests (Tiberi et al. 2016). High-intensity large fires are function of the fuel connectivity both at the stand (Fernandes 2009) and landscape (Fernandes et al. 2016) levels and, therefore, *montados* and *dehesas* can act like fire-friction landscapes in the fullness of their productive functions (Azevedo et al. 2013; Fernandes 2013; Guiomar et al. 2015) since understory management allows the maintenance of low fuel loads. Shrub encroachment is more likely to occur in the more fragmented areas (Acácio et al. 2007; Guiomar et al. 2015), increasing wildfire susceptibility and also the probability of extreme fire behavior under severe weather conditions. Thus, abandoned and fragmented *montados* and *dehesas* are not only less resistant to fire spread, but also less resilient (Guiomar et al. 2015).

As other wood pastures and agroforestry systems throughout Europe, tree aging and recruitment failure must be reversed to guarantee the persistence of these multifunctional landscapes (Bergmeier et al. 2010; Plieninger et al. 2010; Roellig et al. 2018). The traditional "land sparing" strategies are insufficient to reverse the current trend of decline of these multi-functional landscapes and should be complemented by "land sharing" strategies (Grass et al. 2019), to adjust land use intensity by the carrying capacity of the system (García de Jalón et al. 2018). However, landscape-scale approaches must be promoted to define priorities for conservation and operationalise concepts such as the High Nature Value Farming systems. From the biodiversity stand point, it is also essential to monitor the *montado* spatial–temporal dynamics as it supports a large variety of species. To achieve this, the assessment of the relative contribution of the different *montado* patches to their global connectivity, and their evolution over time, is a critical requirement.

Graph theory based approaches (Urban and Keitt 2001; Foltête et al. 2014) have been widely used in connectivity studies to approach distinct topics such as forests, protected areas, wildlife corridors design and ecological restoration (Saura et al. 2011; Pirnat and Hladnik 2016; Santini et al. 2016; Pereira et al. 2017; Hofman et al. 2018; Volk et al. 2018). Not surprisingly, a variety of connectivity related tools has also been developed to fill the researchers' needs (Fuller and Sarkar 2006; Saura and Torné 2009; Foltête et al. 2012; Mestre et al. 2019).

In this study we use graph theory to analyse the multi-temporal *montado* connectivity in the years 1984, 1999 and 2014 with the goal to find out what the changes represent to the short and medium disperser species in terms of habitat availability. Based on the literature, which reveals a *montado* loss trend (Godinho et al. 2016c), we anticipate there has been a loss of habitat affecting many *montado* faunal species and that low mobility species may be particularly affected since their low mobility implies a lower habitat availability. In the end, we intent to provide biodiversity related information useful for future *montado* planning and management.

Methods

Study area

The study area covers 856,720 hectares in Southern Portugal (Fig. 1) (centre of the study area: 388 440 27.6000 N; 78 410 31.2000 W), in a region where *montado* is a typical land use system (Bugalho et al. 2009). The Portuguese *montado* is a multifunctional agro-silvopastoral system, similar to the Spanish *dehesa*, that covers most of the Southern region of the country, Alentejo (Pinto-Correia et al. 2011). The tree cover is composed of cork oaks (*Quercus suber*) and/or holm oaks (*Quercus rotundifolia*) in different densities (Godinho et al. 2016a) and the management practices also shape the understory that may be cultivated or a natural pasture (Canteiro et al. 2011). Thus, the *montado* is a complex system that depends not only on site variability (soil, climate, topography, etc.) but also on the conjunction of production activities (agriculture, livestock, etc.) that share the
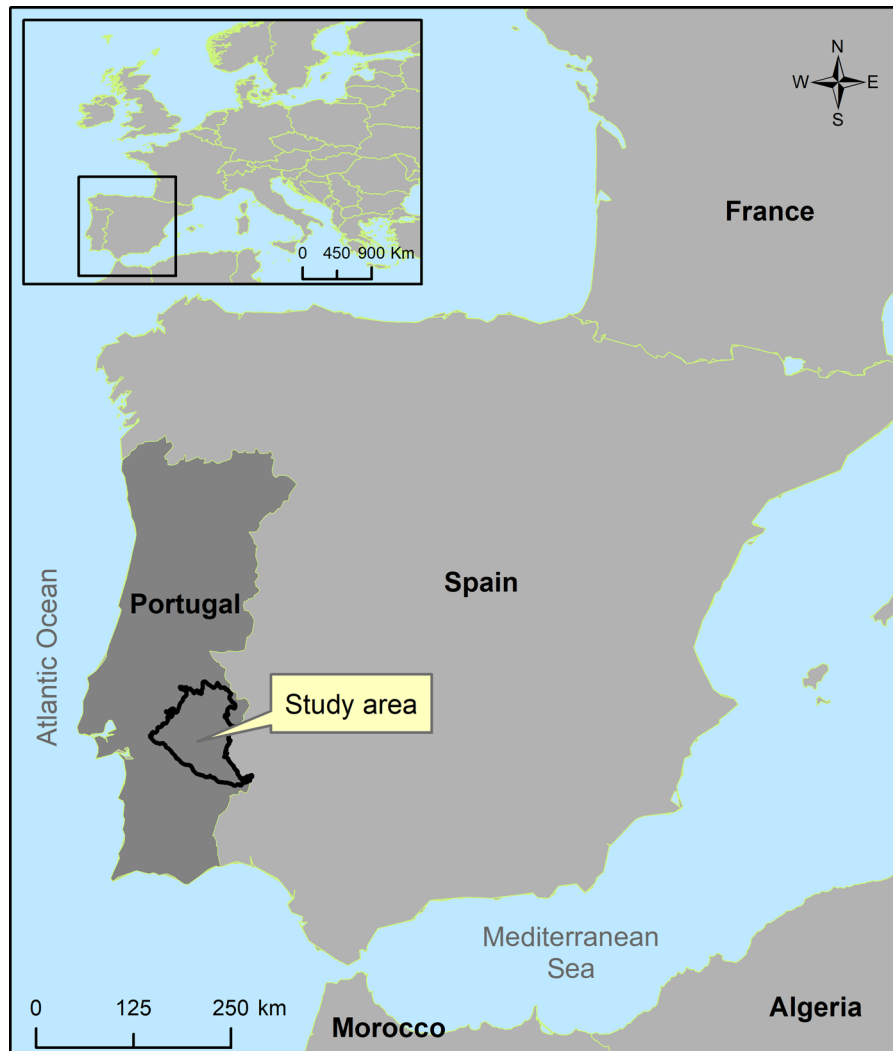
Fig. 1 Study area

same space in the landscape (Pinto-Correia 1993). The system has found economic sustainability for centuries based on a diversity of products such as cork, cereal and livestock, but also complementary products such as wool, firewood and charcoal, among others (Gaspar et al. 2007).

The climate is markedly Mediterranean characterised by hot and dry summers and wet and cold winters. Mean annual precipitation varies between 550 and 650 mm. The elevation ranges from 40 to 645 m and is characterised by soft relief. The study area, selected according to the biogeography boundaries, comprises the Alto Alentejano Superdistrict (Costa et al. 1998). *Montado* covers about 44.8% of the study

area, being the predominant land use system in the region, followed by arable land (27.9%).

Montado land cover maps production

The Landsat program provides the largest temporal records of space-based Earth observations, having been acquiring images of Earth's surface for more than 40 years (Roy et al. 2014). The spatial and spectral resolutions of the multispectral data acquired by the Landsat Thematic Mapper (TM), Enhanced Thematic Mapper Plus (ETM+), and Landsat Operational Land Imager (OLI) make it suitable for mapping and monitoring *montado* ecosystem at a 30 m spatial

resolution since the early eighties. However, producing accurate *montado* maps is a challenging task due to the fuzzy boundaries, in great part caused by tree density variability (Van Doorn and Pinto-correia 2007). Nevertheless, recent works have shown that by combining remote sensing with machine learning algorithms it is possible to produce such maps with fairly good accuracy levels (Godinho et al. 2016a, b; Allen et al. 2018). In this study, satellite imagery from Landsat TM, ETM+, and OLI sensors (path 203 and row 33) were used to produce the *montado* land cover maps for 1984, 1999, and 2014, respectively. For each year, two Landsat scenes of the same study area were acquired, one in spring and the other one in summer, to ensure that inter-class separability benefited from phenological variation of the vegetation cover (Rodriguez-Galiano and Chica-Olmo 2012; Godinho et al. 2016b). Due to the absence of cloud-free Landsat images over the entire spring season of 1984 and 1999, the images from April and June of 1985 and 2000, respectively, were used instead. As a pre-processing step an atmospheric correction was applied to the Blue, Green, Red, NIR, SWIR1 and SWIR2 Landsat bands using the FLAASH (Fast Line-of-sight Atmospheric Analysis of Spectral Hypercubes) method (FLAASH 2009). After atmospheric correction, six vegetation indices were computed for each season as auxiliary variables in image classification procedure: Enhanced Vegetation Index (EVI), Short-Wave Infrared Ratio (SWIR32), Carotenoid Reflectance Index 1 (CRI1), Green Chlorophyll Index (CIgreen), Normalised Multi-band Drought Index (NMDI), and Soil-Adjusted Total Vegetation Index (SATVI)—for more details about the effectiveness in using these indices in semi-arid environments such as the one here addressed see Godinho et al. (2016b). For each year, a layer stack with Landsat bands and derived vegetation indices was produced and classified into eleven representative land cover types: (1) *montado*, (2) pine forest, (3) eucalyptus forest, (4) olive groves, (5) vineyards, (6) irrigation agriculture, (7) dry crops/pastures, (8) shrublands, (9) water bodies, (10) bare soils, and (11) urban areas. Stochastic Gradient Boosting (SGB) algorithm (Friedman 2002) was used to classify Landsat scenes using 1300, 1301, and 1549 sample points (80% for training and 20% for validation) for the years 1984, 1999, and 2014, respectively. Sample data collection was performed based on a stratified approach by land cover types (Table 1) and through

Table 1 List of land cover types and number of sample points used for classification

Class code	Class name	Number of sample points		
		1984	1999	2014
MO	*Montado*	417	417	420
EF	Eucalyptus forest	80	83	117
SL	Shrubland	80	81	81
PF	Pine forest	80	80	80
WT	Water	90	85	89
OG	Olive grove	110	102	148
IA	Irrigation agriculture	80	81	101
C/P	Dry crops/pastures	123	117	213
BS	Bare soil	80	80	81
UB	Urban	80	80	80
VI	Vineyards	80	95	139

Landsat image-interpretation and posterior cross-validation process using different auxiliary georeferenced information; (i) the national Land Cover Map of 1990 (scale 1:25,000) in the case of 1984; (ii) the high-resolution true-color orthophotomaps (0.5 m resolution) produced in 2005 by CNIG—National Center of Geographic Information for the 1999 period; and (iii) the high-resolution imagery from 2011 to 2013 available in Google Earth for the 2014 moment. In addition, for the 1999 and 2014 periods, and taken the advantage of the panchromatic band in ETM+ and OLI sensors, a true-color composition at 15 m spatial resolution of the summer season was produced for improving the photo-interpretation process.

For the accuracy assessment of the land cover products, the validation data set was used to produce the confusion matrix and compute four accuracy measures: overall classification accuracy (OA), producer's accuracy (PA), user's accuracy (UA) and Kappa coefficient (K) (Congalton and Green 2009).

Montado connectivity assessment

In a graph theory framework, landscapes are represented as graphs and their elements correspond to graph elements. In this study, *montado* patches are considered nodes that are surrounded by the non-*montado* land cover that is less suitable or unsuitable for the *montado* species. Distances between *montado* patches associated to the species threshold

dispersal distances (*d*) dictate if a link exists between two nodes. A group of linked nodes forms a connected region, known as component in the graph theory terminology (Pascual-Hortal and Saura 2006). Therefore, a *montado* component encompasses the available habitat for a given specimen because it cannot reach other *montado* components.

Global connectivity

Connectivity analysis was approached via graph theory using the software "Conefor 2.6" (Saura and Torné 2009). The analysis was based on the Integral Index of Connectivity (IIC) (Pascual-Hortal and Saura 2006) that adequately detects different relevant changes in the landscape (Saura and Pascual-Hortal 2007). The IIC is given by:

$$IC = \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} \frac{a_i \cdot a_j}{1+nl_{ij}}}{A_L^2}$$

n—total number of nodes in the landscape; a_i and a_j—attributes of nodes *i* and *j*; nl_{ij}—number of links (shortest path) between patches *i* and *j*; A_L—maximum landscape attribute.

When the nodes are representing habitat patches and the attribute is the area then A_L is the total area of the analysed region, habitat and non-habitat. IIC ranges between 0 (no habitat present in the study area) and 1 (all the study area covered by habitat). Although our goal was to assess how connectivity changed over time, the study did not focus on a particular species. Instead, IIC was calculated for each date using several *d*: 50 m, 100 m, 200 m, 300 m, 500 m, 1500 m and 2000 m. This wide range of *d* encompasses a large variety of important species for biodiversity conservation in the *montado* ecosystems (Sutherland et al. 2000; Martín-Martín et al. 2013), such as small and medium mammals (Rosalino et al. 2009), reptiles (Godinho et al. 2011), amphibians (Ferrand de Almeida et al. 2001) and passerines (Godinho and Rabaça 2011).

The parameter *d*, that is not present in the IIC expression but instead is provided to the software during the analysis, highly affects the IIC because it has direct implication in the number of links.

Individual patch importance

The relative importance of each patch to the global connectivity is given by the dIIC. The values are obtained by simulating a change in the network, such as the removal of a certain patch (Pascual-Hortal and Saura 2006). dIIC is given by:

$$dIIC = 100 \cdot \frac{IIC - IIC'}{IIC}$$

where IIC is the value of the global connectivity before the change and IIC' the value of the same index after the change. We used dIIC values to produce maps showing the different patches contribution to the global connectivity for the three dates. This task was performed using a conservative *d* = 200 m because species with moderate dispersal abilities are more likely to be affected by landscape changes related to connectivity (Saura and Rubio 2010).

IIC is constituted by three partial indices, IIC_{intra}, IIC_{flux} and $IIC_{connector}$. The first represents the intra patch connectivity, the second considers the fluxes in the landscape (inter patch connectivity) and the third calculates the importance of each patch in keeping others patches connected. Derived from these, there are $dIIC_{intra}$, $dIIC_{flux}$ and $dIIC_{connector}$, which represent the partial contribute of each patch to these primary indices (Saura and Rubio 2010).

$$dIIC = dIIC_{intra} + dIIC_{flux} + dIIC_{connector}$$

Top component analysis

A component is a patch or group of patches isolated from the other patches of the same land cover. By definition, an individual lives in one component (single or multi-patch) and is unable to reach other patches of the same habitat. In this work, the term "habitat" should be understood as equivalent to the land cover type *montado*. We calculated the components for the different dates using *d* = 50 m, 100 m, 200 m, 300 m, and 500 m, in an attempt to assess how the *montado* availability/reachability varies for species with different moving abilities. Next, we focused on the five larger components, analysed them in terms of area and number of patches (NP) and produced maps showing their spatiotemporal dynamics for *d* = 200 m.

Results and discussion

Montado land cover maps production

The performed SGB classification using multi-seasonal Landsat spectral bands and selected vegetation indices showed an overall moderate agreement and good accuracy for all the years: 1984 (OA = 81.85%; K = 0.78), 1999 (OA = 75.58%; K = 0.71), and 2014 (OA = 80.07%; K = 0.77). Regarding the *montado* areas, the SGB classification also produced reasonable accuracies: 1984 (PA = 89.2%; UA = 74.7%), 1999 (PA = 91.6%; UA = 80.0%), and 2014 (PA = 85.7%; UA = 75.8%). Based on the confusion matrix computed for each year (see Supplementary material), it can be seen that some areas of *montado* were misclassified as olive groves, vineyards and dry crops/pastures, and vice versa. As outlined in Godinho et al. (2016b), these errors occurred due to the spatial variability in tree density in *montado* ecosystem, as well as the sparse nature of the vegetation cover in olive groves and vineyards. Specifically, in low-density *montado* areas (tree cover between 10 and 30%), olive groves and also in vineyards, the high reflectance from the understory vegetation and soils can overwhelm the reflected components of the sparse canopies contributing to a lower inter-class separability between these land cover types (e.g. Berberoglu et al. 2000, 2009; Fisher et al. 2016). In order to deal with this context-specific nature of the *montado* ecosystem, the use of high spatial, spectral and temporal resolution data such as the ones provided by the Sentinel-2A and 2B sensors may be explored (Godinho et al. 2018). Nevertheless, despite these errors in *montado* areas classification, which were in fact consistent over the three years studied, the comparison between the *montado* maps produced for each year seems acceptable considering the average classification accuracy obtained for 1984 (81.9%), 1999 (85.8%), and 2014 (80.8%). The *montado* patches were then extracted from these original land cover maps and used for the subsequent analysis (Fig. 2).
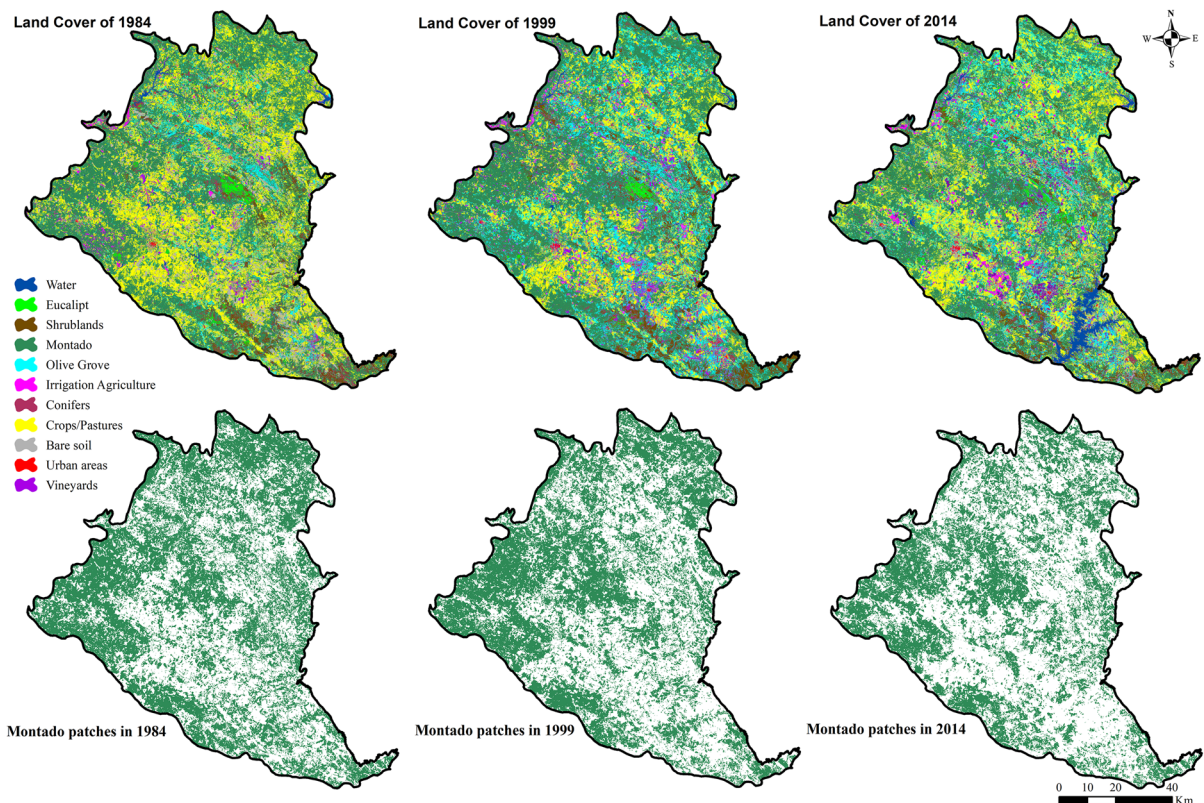

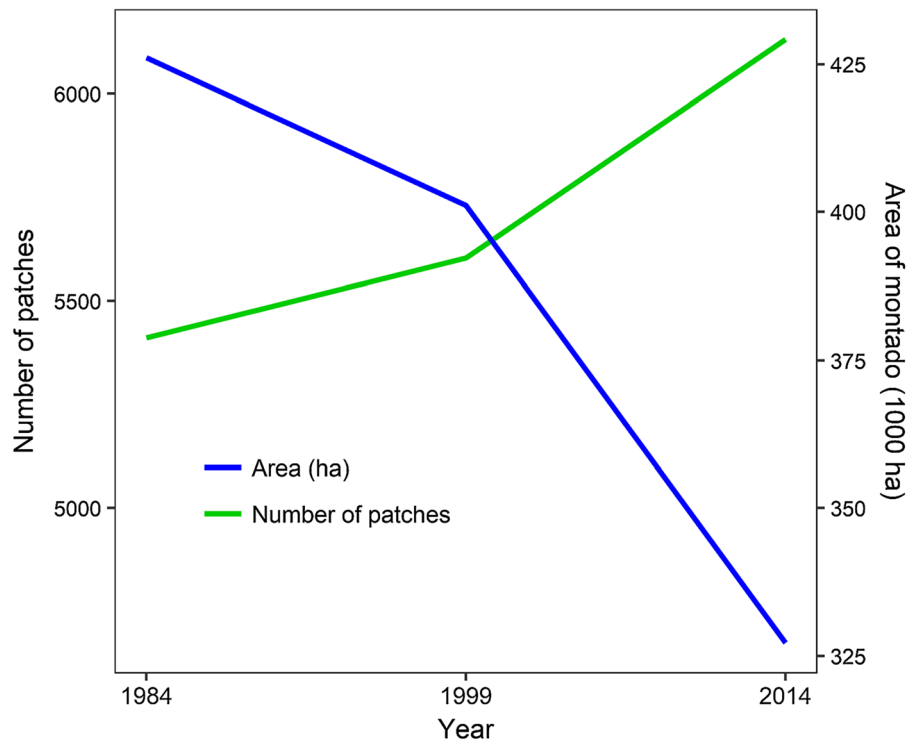**Fig. 2** Land cover and *montado* maps

Fig. 3 *Montado* area and number of patches over time

The total *montado* area in 1984, 1999 and 2014, was respectively 426.062 ha, 401.118 ha and 327.178 ha, therefore totalizing a loss of 98.884 ha. This result is in line with the trend reported by Godinho et al. (2016c) in a study focused on *montado*, using distinct methods and a larger study area. Regarding NP, it showed an opposite trend, increasing from 5411 to 5603 to 6131, resulting in an increment of 720 patches (Fig. 3). Considering partial and overall changes, the trends are the same: *montado* area decreased and the number of *montado* patches increased. The area loss reveals a change in the *montado* composition (how much there is) and the NP increment shows a change in the *montado* configuration (how it is distributed). The combination of both suggests the existence of *montado* fragmentation due to area loss (Machado et al. 2018) at the landscape scale.

Montado connectivity analysis

Global connectivity

Montado connectivity as a whole, expressed by IIC, has been decreasing over time. The 1984 year values

are higher than those of 1999 which are higher than those of 2014 (Fig. 4). Probable reasons for "fragmentation by loss" dynamics having caused lower connectivity are (i) the *montado* loss direct influence in the IIC_{intra} and (ii) the fragmentation's potential contribution to the loss of links, mainly for smaller *d*. The results also show that connectivity increases with the increment of *d*, which is expected because the amount of available *montado* depends on the species dispersal ability. More mobile species, able to cross non-*montado* areas to reach other *montado* patches, use a larger portion of the existing *montado* in the landscape (Saura and Rubio 2010). Results show that IIC for species with *d* = 500 m, was around four times less in 2014 ($\approx$ 0.1) compared to 1984 ($\approx$ 0.025). Even species with $d \approx$ 2000 m suffered similar impacts in what general connectivity is concerned (Fig. 4).

Individual patch importance

Focusing on the patches individually allows us to assess which ones contribute the most to preserve connectivity (Jordán et al. 2003). The dIIC changes
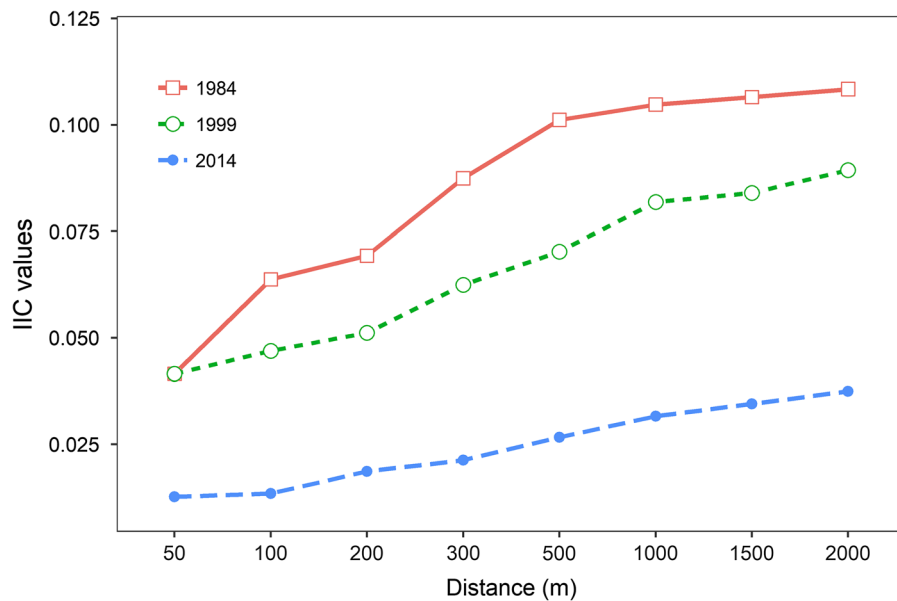
Fig. 4 Global connectivity (IIC) for different *d* over time

indicate the patches' relative importance to the global connectivity has varied over time. Particularly, the multi-temporal analysis using *d* = 200 m (Fig. 5) showed a noteworthy variability in the importance of some patches and makes clear that the "fragmentation by loss" occurred in the *montado* led to a reduction of its connectivity. Some resilient patches that have been fundamental to preserve the global connectivity, and kept high importance over time, have been losing area. This trend is particularly concerning given the amount of Natura 2000 sites present in the study area. See for instance the loss of importance of the *montado* in the "Monfurado" and "São Mamede" Special Areas of Conservation (PTCON0031 and PTCON0007, respectively) and "Évora" Special Protection Area (PTZPE0055).

Once a patch can contribute to connectivity with several functions and play different roles in the network (Estrada and Bodin 2008; Saura and Rubio 2010) we must go beyond dIIC in order to get more in-depth information. A patch can be important due to its size (provides intra connectivity), because its location facilitates fluxes in the network (provides inter connectivity) and/or because it connects two or more patches or groups of patches that would otherwise be disconnected (act as connector). The maps in Fig. 6 show the evolution of $dIIC_{intra}$, $dIIC_{flux}$ and $dIIC_{connector}$ over time. The variations of all indices in the

years 1984, 1999 and 2014 are evident. Intra connectivity ($dIIC_{intra}$) depends only on the patch size and not on *d* or how the patch is connected to other patches (Saura and Rubio 2010). For that reason, it reflects directly the *montado* amount and changes in patch size. Area gain originates higher $dIIC_{intra}$ and area loss originates lower $dIIC_{intra}$. Intra connectivity showed moderate relevance in the central/western part of the study area in 1984, increased its importance until 1999, and kept high values although losing area by 2014.

Inter connectivity ($dIIC_{flux}$) depends on the patch area and on its position within the network (Saura and Rubio 2010). Therefore, changes result from both *montado* composition and configuration. This means not only the quantitative dynamics (loss and gain) of *montado* are important but also the geometric dynamics (NP increment or reduction as each patch position is relevant for establishing links/paths). Inter connectivity was very relevant in the central, western and northern zones of the study area in 1984. After, it lost some of its value in the centre and west, and most of it in the north (1999). Finally, in 2014, it is possible to see some recovery of the value but a loss of area is also noticeable.

$dIIC_{connector}$ depends exclusively on the topological position of a patch in the landscape network (Saura and Rubio 2010). Patch area is irrelevant for this index
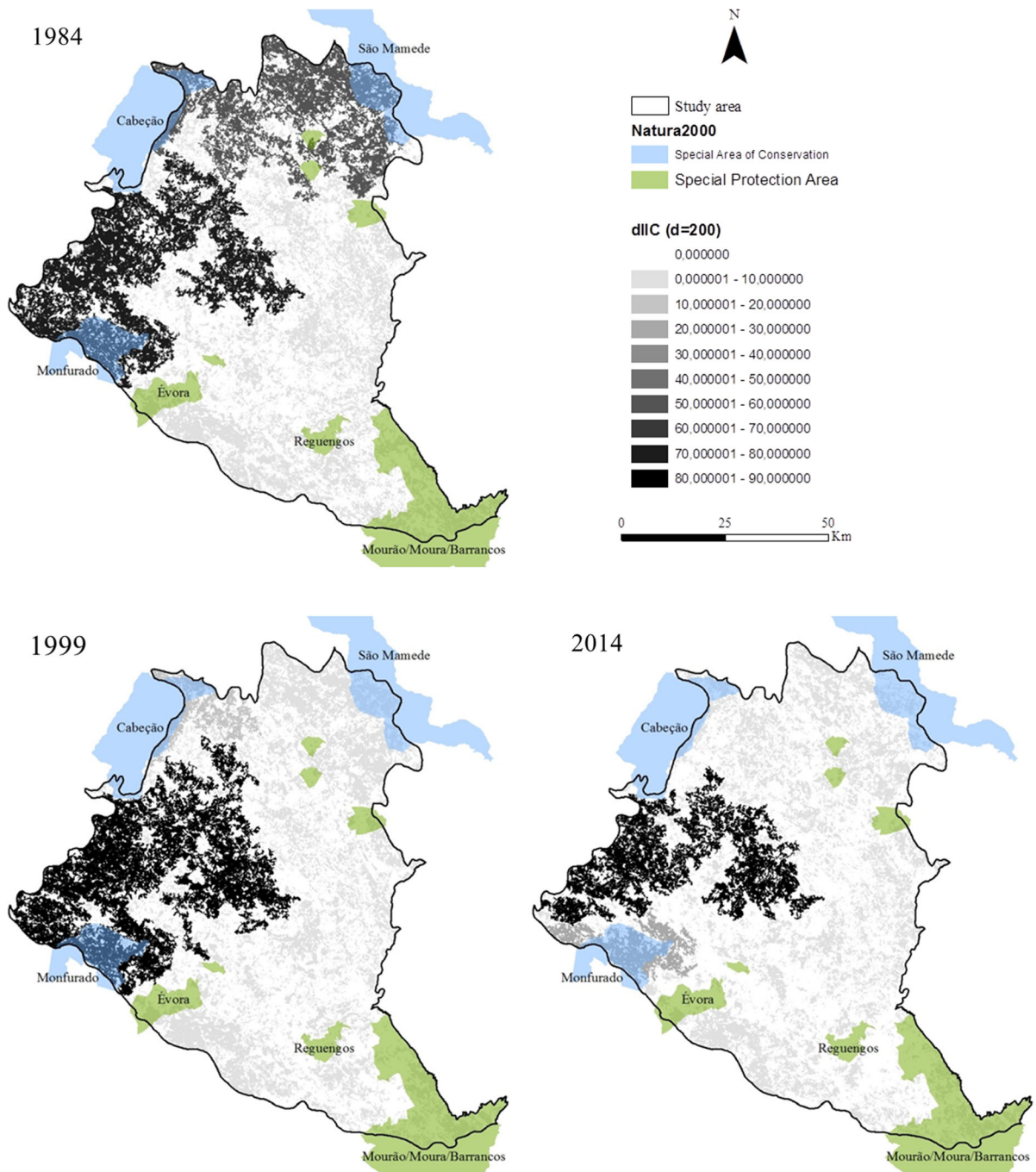
Fig. 5 dIIC maps for $d = 200$ m

that reflects only the landscape configuration dynamics. In this context, changes are consequence of increasing or decreasing *montado* NP due to the role some of them may play as stepping-stones keeping the components' cohesion. dIIC$_{connector}$ is the fraction

with the most variations of all three (Fig. 6). In 1984, it was highly important in the northern part of the study area and moderately important in the centre and west. In 1999, it had lost most of the importance in the north and kept stable in the centre/west. By 2014, it had

Fig. 6 dIIC$_{intra}$, dIIC$_{flux}$ and dIIC$_{connector}$ maps for a $d = 200$ m
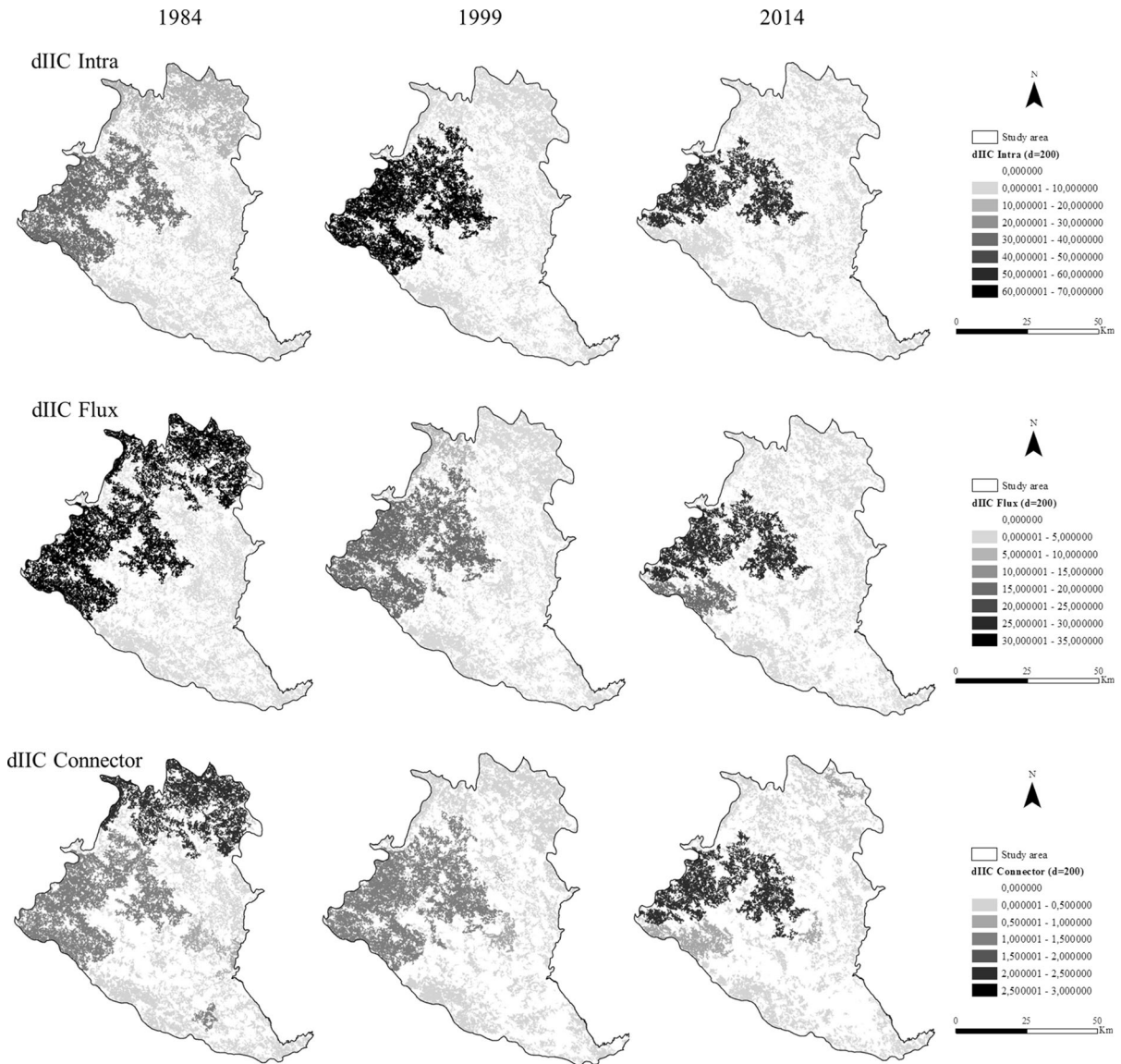
increased its value although its area decreased. These variations are expected to have significant impacts in the populations because the connector patches are directly implied in habitat availability as their existence or inexistence determines more or less habitat at disposal (Gurrutxaga et al. 2011). In a scenario of habitat loss and increasing fragmentation, the connectivity between patches becomes even more important because it may be the only way in which enough habitat can be made available for populations (Jordán 2001).

The importance of knowing the contribute of each patch to the global connectivity is clearly demonstrated in the work by Dondina et al. (2018). Their study aimed at assessing connectivity in an agroecosystem in the Lombardy region (Italy) and was focused on the hazel dormouse. By comparing actual and simulated land covers, the authors found that establishing hedgerows along new pathways would promote higher connectivity gains than restoring/improving existing corridors. In that particular case, instead of trying to enhance relatively less important

elements (with low dIIC, for instance), it would be more beneficial to promote new ones.

Top component analysis

While dIIC analysis works at the patch level and requires deeper understanding of the underlying processes, landscape components analysis is a more expedite and simple way to extract valuable information (Neel et al. 2014). As previously mentioned a component is a connected region and represents the maximum habitat available for a given individual (Pascual-Hortal and Saura 2006). Larger components are expected to harbour more species than smaller ones as they provide more suitable habitat, are less prone to stochastic extinctions and thus are more likely to maintain viable populations (MacArthur and Wilson 1967; Lande 1988). Besides the size, the number of components (NC) also matters because it is directly related to connectivity and habitat availability. For a given area, more components are synonym of discontinuity and less available habitat.

It is important to clarify that NC itself, is not good or bad. For instance, a *montado* area gain located far away so that it would remain disconnected from the network, (e.g. a new plantation) originating a new patch, would form a new component that would be understood as beneficial. However, in our case study we know the *montado* area has been decreasing and the dynamics occurring is "fragmentation by loss" rather than "number of patch increment by gain" (Godinho et al. 2016c; Machado et al. 2018).

The numbers of components for different d in each moment are shown in Fig. 7a. Naturally, the values decrease as the d increase because species with a larger d can reach more patches than a species with a smaller d. For the former, there will be fewer components in the landscape because there are fewer unreachable patches. Conversely, for the latter there are more unreachable patches and thus, more components. The values of each d increased over time reflecting the increasing number of isolated patches and consequent lower connectivity. This also means that many components are composed by a single patch with virtually no influence in the *montado* connectivity.

The overall *montado* area reduction is noticeable when we look at the five larger components. These constitute the core structure of the *montado* and comprise most of the available area for species with

$d > 200$ m (Fig. 7b). For lower d (50 m and 100 m) this is also verifiable for 1984 and 1999. By 2014, changes in the *montado* had been so severe that more components were needed to comprise a higher percentage of *montado* area. For higher d (300 m and 500 m) the values remain similar along the years because the species are mobile enough to reach *montado* patches despite the loss of area occurred. This means the majority of the remaining *montado* patches are closer than 300 m.

A comparable analysis considering the NP percentage instead of area percentage has similar results showing that for a given d, the NP accessible has been decreasing over time (Fig. 7c). Species capable of movements of 300 m and higher have access to almost all the *montado* patches in the landscape while species with very limited moving abilities (50 m or 100 m) only have access to a small percentage of patches.

A more detailed analysis for $d = 200$ m shows how the available *montado* area has decreased over time. It's also noticeable the five larger components have experienced little stability during the studied period (Fig. 7d). Between 1984 and 1999, the top 5 components lost 68,812 ha (401,362–332,550 ha). Between 1999 and 2014, the top 5 components lost 104,424 ha (332,550–228,126 ha). Between 1984 and 2014, the top 5 components lost a total of 173,236 ha (401,362–228,126 ha) (Fig. 7c).

The maps in Fig. 8 show how the five largest components changed during the studied period for $d = 200$ m. Taking in consideration that one individual can only move within one component it is clear the habitat has become more fragmented and less accessible. For instance, a population living in the largest (green) component in 1984 had its component split in two by 1999 (green and yellow), which means those patches were no longer linked and consequently the population was divided.

Which species or group of species are the most affected by the increasing number of *montado* components?

First of all, it is important to stress the *montado* biological value does not come from a single species, nor there are any exclusive species to this land use system. Instead, its richness in terms of biodiversity results from the quantity of species it harbours. Thus, changes in the *montado*, such as splitting components, more than affecting a single species, are expected to provoke changes in the species pool. Any species
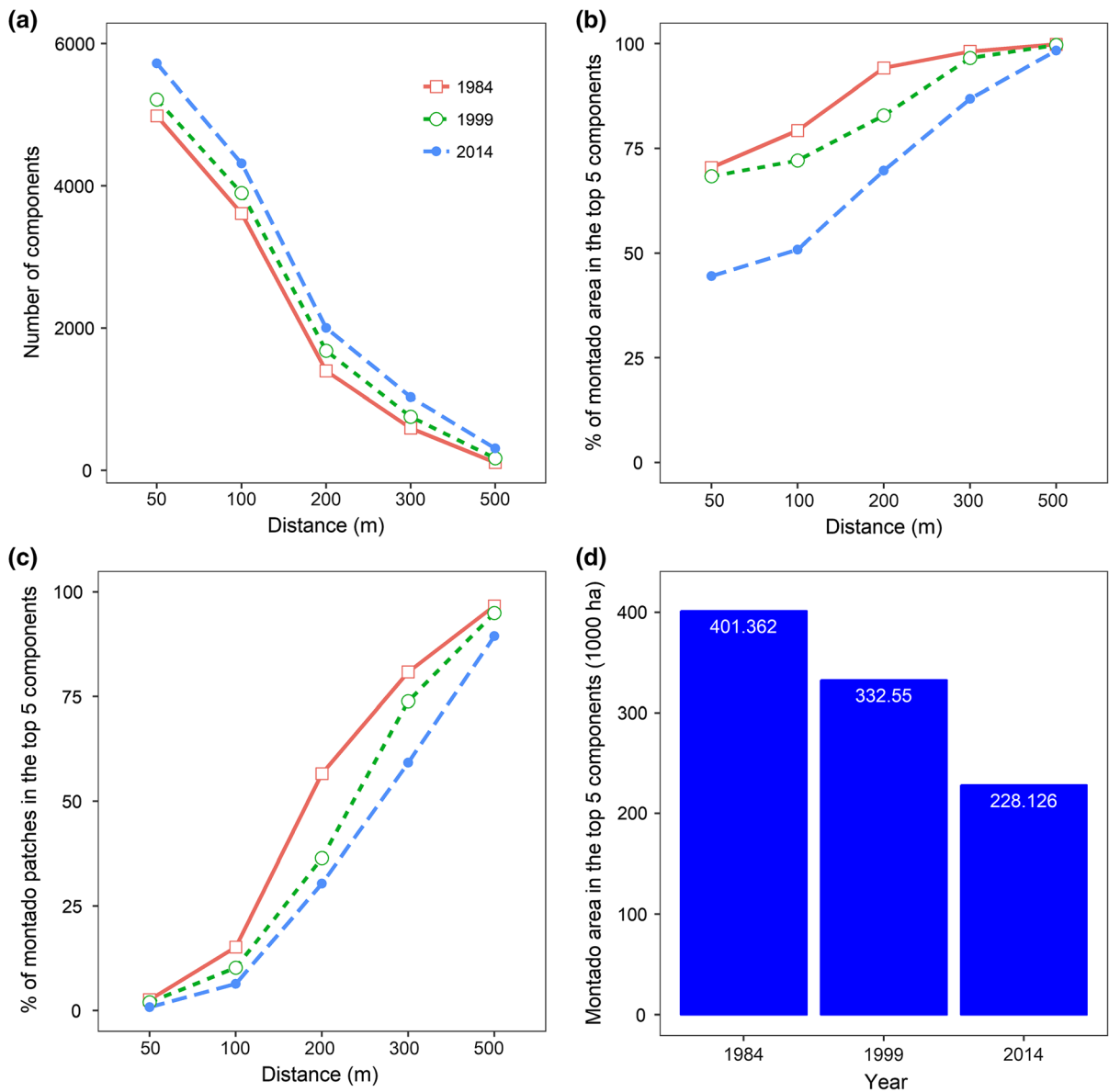
Fig. 7 *Montado* components in 1984, 1999 and 2014; **a** Number of components for different dispersal distances (*d*); **b** *Montado* area (%) in the top 5 components for different *d*; **c** *Montado* patches (%) in the top 5 components for different *d*; **d** *Montado* area in the top 5 components for *d* = 200 m

whose habitat is reduced and disconnected is affected to some extent. In this particular analysis the resulting components show how fragmented the *montado* has become for species with d ≤ 500 m (Fig. 7). Many small mammals belong to these group, among them the *Apodemus sylvaticus* (Dickman and Doncaster (1989)) that favours oak recruitment by spreads acorns (Muñoz and Bonal 2007).

Many species live in the *montado* because the system fits their habitat requirements but do not necessarily depend on it. Others, more specialized and less adaptable display a higher degree of dependence. Focusing on birds as an example, farmland specialists and forest specialists occur only in certain farmlands and forests, respectively. Common farmland birds can occur in a variety of open lands and the common forest birds can live in several forest types. There are also
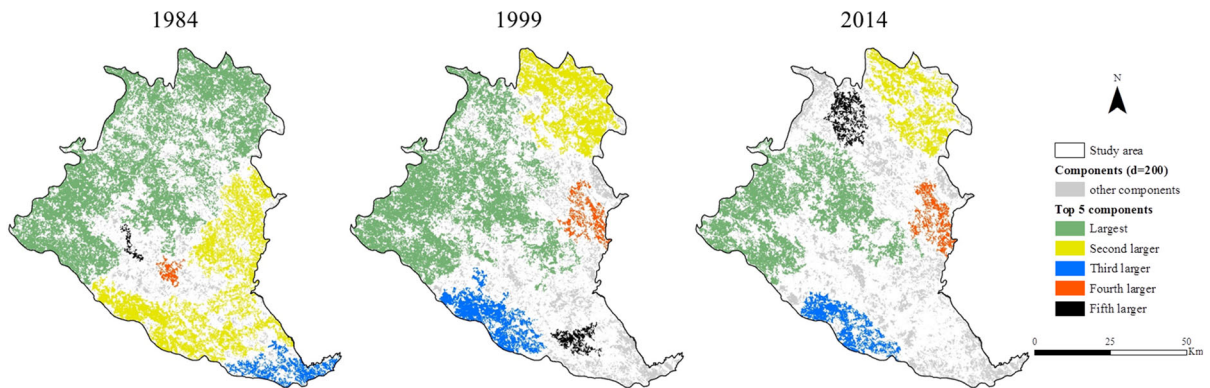
Fig. 8 Spatiotemporal dynamics of the *montado* top 5 components for *d* = 200 m. Larger components by size: 1st—green, 2nd—yellow, 3rd—blue, 4th—red, 5th—black, remaining components—light grey

transition habitat birds that thrive in the interface of open lands and forests. The *montado* structure with pastures and different shrub and tree densities makes it a land use type often used by all of the above groups, except some farmland specialists that completely avoid the presence of trees (Pereira and Godinho 2015; Catarino et al. 2016). Although *montado* degradation can be harmful to many, the forest specialists who live in the *montado* are the most dependent on the system stability and thus are the most affected by a level of loss and/or fragmentation that breaks *montado* components apart. In our study area, *Sitta europaea* and *Dryobates minor* are examples of species well adapted to mature *montados* that are likely to be more affected by the increasing NC.

Conclusion

The *montado* classification as high nature value farming system helped to raise awareness to its importance but the heterogeneity that differentiates the *montado* has not been reflected in the policies. Several policy instruments affect the *montado* and have allowed its economic valorisation but do not consider it as a whole, compromising the survival of many *montados* in the medium-term (Pinto-Correia et al. 2013). Concrete measures are needed that effectively improve multi-functional farming systems and support practices that preserve biodiversity such as extensive grazing in high nature value farmlands (Pe'er et al. 2020).

The biodiversity aspect is particularly relevant in the *montado* because the system's well-functioning

and sustainability is highly dependent on natural processes. Without proper biological support, *montado* becomes a poor and unsustainable system threatened by abandonment. Thus, it is essential to guarantee suitable conditions for wildlife to prosper and for that connectivity plays a key role.

Montado changes and its resulting connectivity in the years 1984, 1999 and 2014 were assessed based on accurate information derived from satellite remote sensing multispectral data. In this period:

- *Montado* experienced a deep regression in terms of area amount (− 98.884 ha) and an increase in the fragmentation level (+ 720 patches).
- The global connectivity decreased for all the tested *d*, resulting in a negative impact for several species.
- The relative importance of each *montado* patch to the global connectivity varied over time due to changes in spatial composition and configuration.
- The *montado* area represented by the top 5 components varied significantly and the amount displays a decreasing trend. This implies that species with dispersal abilities up to 500 m in 1984 had more *montado* connected areas than in 1999 and 2014. Ultimately, it means the populations are increasingly more isolated in 1999 and 2014.

The most affected species by habitat loss and/or fragmentation are those with low mobility that cannot reach other viable patches via dispersal and those that display higher levels of specialization and thus are more dependent on the habitat's characteristics. In the *montado*, the former are mainly micromammals and

the latter some forest specialist birds such as the *Sitta europaea* and the *Dryobates minor*.

Every *montado* patch (and component) is important but the larger ones are more likely to display higher specific richness, seem to be more resilient and provide more intraconnectivity that is key to short dispersal species, unable or unlikely to move successfully among suitable patches in a fragmented landscape. Hence, at the landscape scale, the priority should be to preserve the core areas that have been the support of connectivity (the larger components) and avoid as much as possible the loss of more area that will consequently provoke fragmentation. A more fragmented montado (higher NC) can also mean that more patches need to be preserved to guarantee high habitat availability and therefore making conservation prioritization harder (Pereira 2018). Biodiversitywise, *montado* has value as a whole and given also its socio-economic importance measures should be taken in order to preserve it as much as possible. By reporting the *montado* decreasing trend and highlighting some of the associated impacts on biodiversity, on which the *montado* itself depends upon, our work also contributes to raise awareness and reinforce the need to preserve and increment this unique land use system.

References

Acácio V, Dias FS, Catry FX, Rocha M, Moreira F (2017) Landscape dynamics in Mediterranean oak forests under global change: understanding the role of anthropogenic and environmental drivers across forest types. Glob Chang Biol 23:1199–1217. https://doi.org/10.1111/gcb.13487

Acácio V, Holmgren M, Jansen PA, Schrotter O (2007) Multiple recruitment limitation causes arrested succession in Mediterranean cork oak systems. Ecosystems 10:1220–1230

Aguilar R, Ashworth L, Galetto L, Aizen MA (2006) Plant reproductive susceptibility to habitat fragmentation: review and synthesis through a meta-analysis. Ecol Lett 9:968–980

Allen H, Simonson W, Parham E, Santos E, Hotham P (2018) Satellite remote sensing of land cover change in a mixed agro-silvo-pastoral landscape in the Alentejo, Portugal. Int J Remote Sens 39:4663–4683. https://doi.org/10.1080/01431161.2018.1440095

Almeida M, Azeda C, Guiomar N, Pinto-Correia T (2016) The effects of grazing management in montado fragmentation and heterogeneity. Agrofor Syst 90:69–85

Arosa ML, Bastos R, Cabral JA, Freitas H, Costa SR, Santos M (2017) Long-term sustainability of cork oak agro-forests in the Iberian Peninsula: a model-based approach aimed at supporting the best management options for the montado conservation. Ecol Model 343:68–79. https://doi.org/10.1016/j.ecolmodel.2016.10.008

Astudillo PX, Schabo DG, Siddons DC, Farwig N (2019) Patch-matrix movements of birds in the páramo landscape of the southern Andes of Ecuador. Emu 119:53–60

Azevedo JC, Possacos A, Aguiar CF, Amado A, Miguel L, Dias R, Loureiro C, Fernandes PM (2013) The role of holm oak edges in the control of disturbance and conservation of plant diversity in fire-prone landscapes. For Ecol Manag 297:37–48. https://doi.org/10.1016/j.foreco.2013.02.007

Berberoglu S, Lloyd CD, Atkinson PM, Curran PJ (2000) The integration of spectral and textural information using neural networks for land cover mapping in the Mediterranean. Comput Geosci 26:385–396

Berberoglu S, Satir O, Atkinson PM (2009) Mapping percentage tree cover from Envisat MERIS data using linear and nonlinear techniques. Int J Remote Sens 30:4747–4766

Bergmeier E, Petermann J, Schröder E (2010) Geobotanical survey of wood-pasture habitats in Europe: diversity, threats and conservation. Biodivers Conserv 19:2995–3014

Blondel J (2006) The 'design' of Mediterranean landscapes: a millennial story of humans and ecological systems during the historic period. Hum Ecol 34:713–729

Bugalho MN, Caldeira MC, Pereira JS et al (2011) Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Front Ecol Environ 9:278–286

Bugalho M, Plieninger T, Aronson J, Ellatifi M, Crespo DG (2009) Open woodlands: a diversity of uses (and overuses). In: Aronson J, Pereira JS, Pausas JG (eds) Cork oak woodlands on the edge, ecology, adaptive management, and restoration, 1st edn. Society for Ecological Restoration International, Island Press, Washington D.C., pp 33–45

Camilo-Alves C, Clara MIE, Ribeiro NMCA (2013) Decline of Mediterranean oak trees and its association with *Phytophthora cinnamomi*: a review. Eur J For Res 132:411–432

Camilo-Alves C, Vaz M, Clara MIE, Ribeiro NMCA (2017) Chronic cork oak decline and water status: new insights. New For 48:753–772

Catarino L, Godinho C, Pereira P, Luís A, Rabaça JE (2016) Can birds play a role as high nature value indicators of montado system? Agrofor Syst 90:45–56. https://doi.org/10.1007/s10457-014-9761-y

Canteiro C, Pinto-Cruz C, Simões MP, Gazarini L (2011) Conservation of Mediterranean oak woodlands: understorey dynamics under different shrub management. Agrofor Syst 82:161–171

Collinge SK (1998) Spatial arrangement of habitat patches and corridors: clues from ecological field experiments. Landsc Urban Plan 42:157–168

Congalton RG, Green K (2009) Assessing the accuracy of remotely sensed data: principles and practices, 2nd edn. CRC Press, Boca Raton, Florida

Costa JC, Aguiar C, Capelo JH, Lousã M, Neto C (1998) Biogeografia de Portugal Continental. Quercetea 1:5–56

Dickman CR, Doncaster CP (1989) The ecology of small mammals in urban habitats. II. Demography and dispersal. J Anim Ecol 58:119

Dinis C, Surový P, Ribeiro N, Oliveira MRG (2015) The effect of soil compaction at different depths on cork oak seedling growth. New For 46:235–246

Dondina O, Saura S, Bani L, Mateo-Sánchez MC (2018) Enhancing connectivity in agroecosystems: focus on the best existing corridors or on new pathways? Landsc Ecol 33:1741–1756

Estrada E, Bodin Ö (2008) Using network centrality measures to manage landscape connectivity. Ecol Appl 18:1810–1825

Eycott AE, Stewart GB, Buyung-Ali LM, Bowler DE, Watts K, Pullin AS (2012) A meta-analysis on the impact of different matrix structures on species movement rates. Landsc Ecol 27:1263–1278. https://doi.org/10.1007/s10980-012-9781-9

Fahrig L, Merriam G (1985) Habitat patch connectivity and population survival. Ecology 66:1762–1768

Farwig N, Bailey D, Bochud E, Herrmann JD, Kindler E, Reusser N, Schüepp C, Schmidt-Entling MH (2009) Isolation from forest reduces pollination, seed predation and insect scavenging in Swiss farmland. Landsc Ecol 24:919–927. https://doi.org/10.1007/s10980-009-9376-2

Fernandes PM (2009) Combining forest structure data and fuel modelling to classify fire hazard in Portugal. Ann For Sci 66:415–415

Fernandes PM (2013) Fire-smart management of forest landscapes in the Mediterranean basin under global change. Landsc Urban Plan 110:175–182

Fernandes PM, Monteiro-Henriques T, Guiomar N, Loureiro C, Barros AMG (2016) Bottom-up variables govern large-fire size in Portugal. Ecosystems 19:1362–1375. https://doi.org/10.1007/s10021-016-0010-2

Ferrand de Almeida N, Ferrand de Almeida P, Gonçalves H, Sequeira F, Teixeira J, Ferrand de Almeida F (2001) Guia FAPAS Anfíbios e Répteis de Portugal. FAPAS e Câmara Municipal do Porto, Porto

Fisher A, Day M, Gill T, Roff A, Danaher T, Flood N (2016) Large-area, high-resolution tree cover mapping with multi-temporal SPOT5 imagery, New South wales, Australia. Remote Sens. https://doi.org/10.3390/rs8060515

Foltête JC, Clauzel C, Vuidel G (2012) A software tool dedicated to the modelling of landscape networks. Environ Model Softw 38:316–327

Foltête JC, Girardet X, Clauzel C (2014) A methodological framework for the use of landscape graphs in land-use planning. Landsc Urban Plan 124:140–150

French K, Mason TJ, Sullivan N (2011) Recruitment limitation of native species in invaded coastal dune communities. Plant Ecol 212:601–609

Friedman JH (2002) Stochastic gradient boosting. Comput Stat Data Anal 38:367–378

Fuller T, Sarkar S (2006) LQGraph: a software package for optimizing connectivity in conservation planning. Environ Model Softw 21:750–755

García de Jalón S, Graves A, Moreno G, Palma JHN, Crous-Durán J, Kay S, Burgess PJ (2018) Forage-SAFE: a model for assessing the impact of tree cover on wood pasture profitability. Ecol Model 372:24–32. https://doi.org/10.1016/j.ecolmodel.2018.01.017

Gaspar P, Mesías FJ, Escribano M, Rodriguez de Ledesma A, Pulido F (2007) Economic and management characterization of dehesa farms: implications for their sustainability. Agrofor Syst 71:151–162. https://doi.org/10.1007/s10457-007-9081-6

Godinho S, Gil A, Guiomar N, Neves N, Pinto-Correia T (2016a) A remote sensing-based approach to estimating montado canopy density using the FCD model: a contribution to identifying HNV farmlands in southern Portugal. Agrofor Syst 90:23–34. https://doi.org/10.1007/s10457-014-9769-3

Godinho S, Guiomar N, Gil A (2016b) Using a stochastic gradient boosting algorithm to analyse the effectiveness of Landsat 8 data for montado land cover mapping: application in southern Portugal. Int J Appl Earth Obs Geoinf 49:151–162

Godinho S, Guiomar N, Machado R, Santos P, Sá-Sousa P, Fernandes JP, Neves N, Pinto-Correia T (2016c) Assessment of environment, land management, and spatial variables on recent changes in montado land cover in southern Portugal. Agrofor Syst 90:177–192. https://doi.org/10.1007/s10457-014-9757-7

Godinho C, Rabaça JE (2011) Birds like it Corky: the influence of habitat features and management of "montados" in breeding bird communities. Agrofor Syst 82:183–195

Godinho S, Santos AP, Sá-Sousa P (2011) Montado management effects on the abundance and conservation of reptiles in Alentejo, Southern Portugal. Agrofor Syst 82:197–207

Godinho S, Surový P, Sousa A, Gil A (2018) Advances in remote-sensing applications in silvo-pastoral systems. Int J Remote Sens 39:4565–4571

Gómez-Aparicio L, Zamora R, Gómez JM, Hódar JA, Castro J, Baraza E (2004) Applying plant facilitation to forest restoration: a meta-analysis of the use of shrubs as nurse plants. Ecol Appl 14:1128–1138. https://doi.org/10.1890/03-5084

Gómez JM (2003) Spatial patterns in long-distance dispersal of Quercus ilex acorns by jays in a heterogeneous landscape. Ecography (Cop) 26:573–584

Grass I, Loos J, Baensch S, Batáry P, Librán-Embid F, Ficiciyan A, Klaus F, Riechers M, Rosa J, Tiede J, Udy K, Westphal C, Wurz A, Tscharntke T (2019) Land-sharing/-sparing connectivity landscapes for ecosystem services and biodiversity conservation. People Nat 3:21. https://doi.org/10.1002/pan3.21

Guiomar N, Godinho S, Fernandes PM, Machado R, Neves N, Fernandes JP (2015) Wildfire patterns and landscape changes in Mediterranean oak woodlands. Sci Total Environ 536:338–352. https://doi.org/10.1016/j.scitotenv.2015.07.087

Gurrutxaga M, Rubio L, Saura S (2011) Key connectors in protected forest area networks and the impact of highways: a transnational case study from the Cantabrian Range to the Western Alps (SW Europe). Landsc Urban Plan 101:310–320

Herrera JM, García D (2010) Effects of forest fragmentation on seed dispersal and seedling establishment in ornithochorous trees. Conserv Biol 24:1089–1098

Hofman MPG, Hayward MW, Kelly MJ, Balkenhol N (2018) Enhancing conservation network design with graph-theory and a measure of protected area effectiveness: refining wildlife corridors in Belize, Central America. Landsc Urban Plan 178:51–59

Ibáñez I, Katz DSW, Peltier D, Wolf SM, Connor Barrie BT (2014) Assessing the integrated effects of landscape fragmentation on plants and plant communities: the challenge of multiprocess-multiresponse dynamics. J Ecol 102:882–895. https://doi.org/10.1111/1365-2745.12223

Johnson AR, Wiens JA, Milne BT, Crist TO (1992) Animal movements and population dynamics in heterogeneous landscapes. Landsc Ecol 7:63–75

Jordán F (2001) Adding function to structure—comments on Palmarola landscape connectivity. Community Ecol 2:133–135

Jordán F, Báldi A, Orci KM, Rácz I, Varga Z (2003) Characterizing the importance of habitat patches and corridors in maintaining the landscape connectivity of a Pholidoptera transsylvanica (Orthoptera) metapopulation. Landsc Ecol 18:83–92. https://doi.org/10.1023/A:1022958003528

Kiviniemi K (2008) Effects of fragment size and isolation on the occurrence of four short-lived plants in semi-natural grasslands. Acta Oecologica 33:56–65

Lande R (1988) Genetics and demography in biological conservation. Science 241:1455–1460

Leiva MJ, Fernández-Alés R (2003) Post-dispersive losses of acorns from Mediterranean savannah-like forests and shrublands. For Ecol Manag 176:265–271

Lopes LE, Buzato S (2007) Variation in pollinator assemblages in a fragmented landscape and its effects on reproductive stages of a self-incompatible treelet, Psychotria suterella (Rubiaceae). Oecologia 154:305–314

MacArthur RH, Wilson EO (1967) The theory of island biogeography. Princeton University Press, Princeton, NJ

Machado R, Godinho S, Pirnat J, Neves N, Santos P (2018) Assessment of landscape composition and configuration via spatial metrics combination: conceptual framework proposal and method improvement. Landsc Res 43:652–664. https://doi.org/10.1080/01426397.2017.1336757

Martín-Martín C, Bunce RGH, Saura S, Elena-Rosselló R (2013) Changes and interactions between forest landscape connectivity and burnt area in Spain. Ecol Indic 33:129–138

Martín Vicente Á, Fernández Alés R (2006) Long term persistence of dehesas. Evidences from history. Agrofor Syst 67:19–28

McEuen AB, Curran LM (2004) Seed dispersal and recruitment limitation across spatial scales in temperate forest fragments. Ecology 85:507–518

Mestre F, Ascensão F, Barbosa AM (2019) gDefrag: a graph-based tool to help defragmenting landscapes divided by linear infrastructures. Ecol Model 392:1–5

Muñoz A, Bonal R (2007) Rodents change acorn dispersal behaviour in response to ungulate presence. Oikos 116:1631–1638

Muñoz A, Bonal R, Díaz M (2009) Ungulates, rodents, shrubs: interactions in a diverse Mediterranean ecosystem. Basic Appl Ecol 10:151–160

Neel M, Tumas HR, Marsden BW (2014) Representing connectivity: quantifying effective habitat availability based on area and connectivity for conservation status assessment and recovery. PeerJ 2:e622

Paracchini ML, Petersen J-E, Hoogeveen Y, Bamps C, Burfield I, can Swaay C (2008) High nature value farmland in Europe—An estimate of the distribution patterns on the basis of land cover and biodiversity data. European Commission, Joint Research Centre, Institute for Environment and Sustainability, Office for Official Publications of the European Communities, Luxembourg

Pascual-Hortal L, Saura S (2006) Comparison and development of new graph-based landscape connectivity indices: towards the priorization of habitat patches and corridors for conservation. Landsc Ecol 21:959–967

Pe'er G, Bonn A, Bruelheide H, Dieker P, Eisenhauer N, Feindt PH, Hagedorn G, Hansjürgens B, Herzon I, Lomba Â, Marquard E, Moreira F, Nitsch H, Oppermann R, Perino A, Röder N, Schleyer C, Schindler S, Wolf C, Zinngrebe Y, Lakner S (2020) Action needed for the EU Common Agricultural Policy to address sustainability challenges. People Nat. https://doi.org/10.1002/pan3.10080

Pereira J (2018) Multi-node protection of landscape connectivity: habitat availability and topological reachability. Community Ecol 19:176–185

Pereira P, Godinho C, Roque I, Rabaça JE (2015) O Montado e as aves: boas práticas para uma gestão sustentável. LabOr – Laboratório de Ornitologia/ICAAM, Universidade de Évora, Câmara Municipal de Coruche, Coruche

Pereira J, Saura S, Jordán F (2017) Single-node vs. multi-node centrality in landscape graph analysis: key habitat patches and their protection for 20 bird species inNESpain. Methods Ecol Evol. https://doi.org/10.1111/2041-210X.12783

Pérez-Ramos IM, Ourcival JM, Limousin JM, Rambal S (2010) Mast seeding under increasing drought: results from a long-term data seat and from a rainfall exclusion experiment. Ecology 91:3057–3068

Pinto-Correia T (1993) Threatened landscape in Alentejo, Portugal: the 'montado' and other 'agro-silvo-pastoral' systems. Landsc Urban Plan 24:43–48

Pinto-Correia T, Guiomar N, Ferraz-de-Oliveira MI, Sales-Baptista E, Rabaça J, Godinho C, Ribeiro N, Sá Sousa P, Santos P, Santos-Silva C, Simões MP, Belo ADF, Catarino L, Costa P, Fonseca E, Godinho S, Azeda C, Almeida M, Gomes L, Lopes de Castro J, Louro R, Silvestre M, Vaz M (2018) Progress in identifying high nature value montados: impacts of grazing on hardwood rangeland biodiversity. Rangel Ecol Manag 71:612–625. https://doi.org/10.1016/j.rama.2018.01.004

Pinto-Correia T, Mascarenhas J (1999) Contribution to the extensification/intensification debate: new trends in the Portuguese montado. Landsc Urban Plan 46:125–131

Pinto-Correia T, Ribeiro N, Potes J (2013) Livro Verde dos Montados. Instituto de Ciências Agrárias e Ambientais Mediterrânicas (ICAAM) - Universidade de Évora, Évora

Pinto-Correia T, Ribeiro N, Sá-Sousa P (2011) Introducing the montado, the cork and holm oak agroforestry system of Southern Portugal. Agrofor Syst 82:99–104

Pirnat J, Hladnik D (2016) Connectivity as a tool in the prioritization and protection of sub-urban forest patches in landscape conservation planning. Landsc Urban Plan 153:129–139

Plieninger T, Rolo V, Moreno G (2010) Large-scale patterns of Quercus ilex, Quercus suber, and Quercus pyrenaica regeneration in central-western Spain. Ecosystems 13:644–660

Puerta-Piñero C (2010) Intermediate spatial variations on acorn predation shapes Holm oak establishment within a Mediterranean landscape context. Plant Ecol 210:213–224

Puerta-Piñero C, Pino J, Gómez JM (2012) Direct and indirect landscape effects on Quercus ilex regeneration in heterogeneous environments. Oecologia 170:1009–1020

Pulido FJ, Díaz M (2005) Regeneration of a Mediterranean oak: a whole-cycle approach. Écoscience 12:92–102

Rodriguez-Galiano V, Chica-Olmo M (2012) Land cover change analysis of a Mediterranean area in Spain using different sources of data: multi-seasonal landsat images, land surface temperature, digital terrain models and texture. Appl Geogr 35:208–218

Roellig M, Costa A, Garbarino M, Hanspach J, Hartel T, Jakobsson S, Lindborg R, Mayr S, Plieninger T, Sammul M, Varga A, Fischer J (2018) Post Hoc assessment of stand structure across European wood-pastures: implications for land use policy. Rangel Ecol Manag 71:526–535. https://doi.org/10.1016/j.rama.2018.04.004

Rosalino LM, Rosário do J, Santos-Reis M (2009) The role of habitat patches on mammalian diversity in cork oak agroforestry systems. Acta Oecol 35:507–512

Rosenweig ML (1995) Species diversity in space and time. Cambridge University Press, Cambridge

Roy DP, Wulder MA, Loveland TR, Woodcock CE, Allen RG, Anderson MC, Helder D Irons JR, Johnson DM, Kennedy R, Scambos TA, Schaafk CB, Schott JR, Sheng Y, Vermote EF, Belward AS, Bindschadler R, Cohen WB, Gao F, Hipple JD, Hostert P, Huntington J, Justice CO, Kilic A, Kovalskyy V, Lee ZP, Lymburner L, Masek JG, McCorkel J, Shuai Y, Trezza R, Vogelmann J, Wynne RH, Zhu Z (2014) Landsat-8: science and product vision for terrestrial global change research. Remote Sens Environ 145:154–172. https://doi.org/10.1016/j.rse.2014.02.001

Santini L, Saura S, Rondinini C (2016) Connectivity of the global network of protected areas. Divers Distrib 22:199–211

Saura S, Estreguil C, Mouton C, Rodríguez-Freire M (2011) Network analysis to assess landscape connectivity trends: application to European forests (1990–2000). Ecol Indic 11:407–416

Saura S, Pascual-Hortal L (2007) A new habitat availability index to integrate connectivity in landscape conservation planning: comparison with existing indices and application to a case study. Landsc Urban Plan 83:91–103

Saura S, Rubio L (2010) A common currency for the different ways in which patches and links can contribute to habitat availability and connectivity in the landscape. Ecography (Cop) 33:523–537

Saura S, Torné J (2009) Conefor Sensinode 2.2: a software package for quantifying the importance of habitat patches for landscape connectivity. Environ Model Softw 24:135–139

Seltmann P, Cocucci A, Renison D, Cierjacks A, Hensen I (2009) Mating system, outcrossing distance effects and pollen availability in the wind-pollinated treeline species Polylepis australis BITT. (Rosaceae). Basic Appl Ecol 10:52–60. https://doi.org/10.1016/j.baae.2007.11.008

Steffan-Dewenter I, Tscharntke T (1999) Effects of habitat isolation on pollinator communities and seed set. Oecologia 121:432–440

Surová D, Ravera F, Guiomar N, Sastre RM, Pinto-Correia T (2018) Contributions of Iberian Silvo-Pastoral landscapes to the well-being of contemporary society. Rangel Ecol Manag 71:560–570. https://doi.org/10.1016/j.rama.2017.12.005

Sutherland GD, Harestad AS, Price K, Lertzman KP (2000) Scaling of natal dispersal distances in terrestrial birds and mammals. Ecol Soc. https://doi.org/10.5751/ES-00184-040116

Taylor PD, Fahrig L, Henein K, Merriam G (1993) Connectivity is a vital element of landscape structure. Oikos 68:571–573

Tiberi R, Branco M, Bracalini M, Croci F, Panzavolta T (2016) Cork oak pests: a review of insect damage and management. Ann For Sci 73:219–232. https://doi.org/10.1007/s13595-015-0534-1

Urban D, Keitt T (2001) Landscape connectivity: a graph-theoretic perspective. Ecology 82:1205–1218

Van Doorn AM, Pinto-correia T (2007) Differences in land cover interpretation in landscapes rich in cover gradients: reflections based on the montado of South Portugal. Agrofor Syst 70:169–183

Volk XK, Gattringer JP, Otte A, Harvolk-Schöning S (2018) Connectivity analysis as a tool for assessing restoration success. Landsc Ecol 33:371–387

With KA, Gardner RH, Turner MG (1997) Landscape connectivity and population distributions in heterogeneous environments. Oikos 78:151–169