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RESEARCH ARTICLE

Supporting the spatial management of invasive alien plants through assessment of landscape dynamics and connectivity

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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

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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

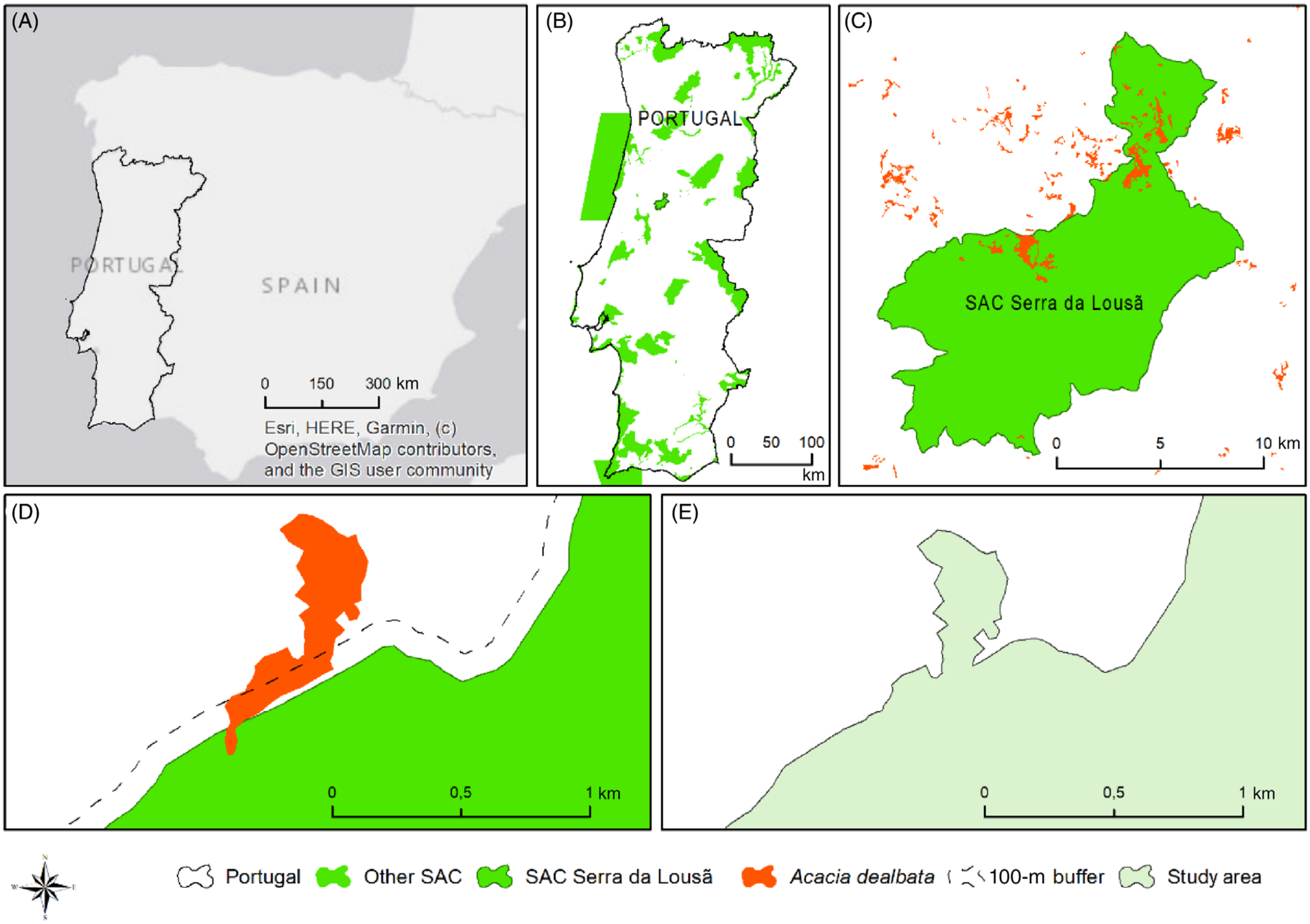


Figure 1. Study area. (A) Iberian Peninsula; (B) Special Areas of Conservation (SAC) in mainland Portugal; (C) SAC "Serra da Lousã" and *Acacia dealbata*; (D) detailed view of the 100-m buffer; (E) incorporation of the buffer and intersected *A. dealbata* patches in the study area.

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).

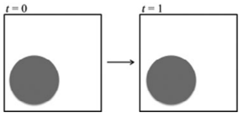
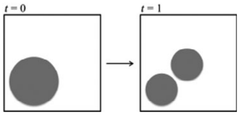
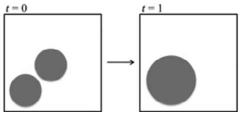
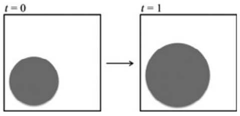
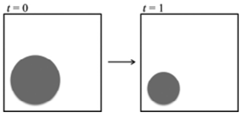
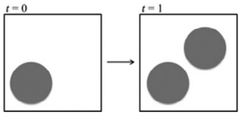
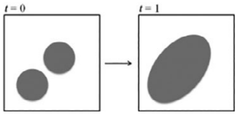
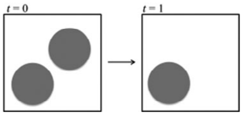
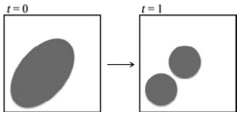
Regarding dispersal processes and patterns, the *A. dealbata* seed bank is mainly located under the tree canopy and its density declines steeply away from canopy (Passos et al. 2017). The seeds are mainly ant-dispersed which also limits their spreading distance (Gibson et al. 2011). Exceptions to these 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)	

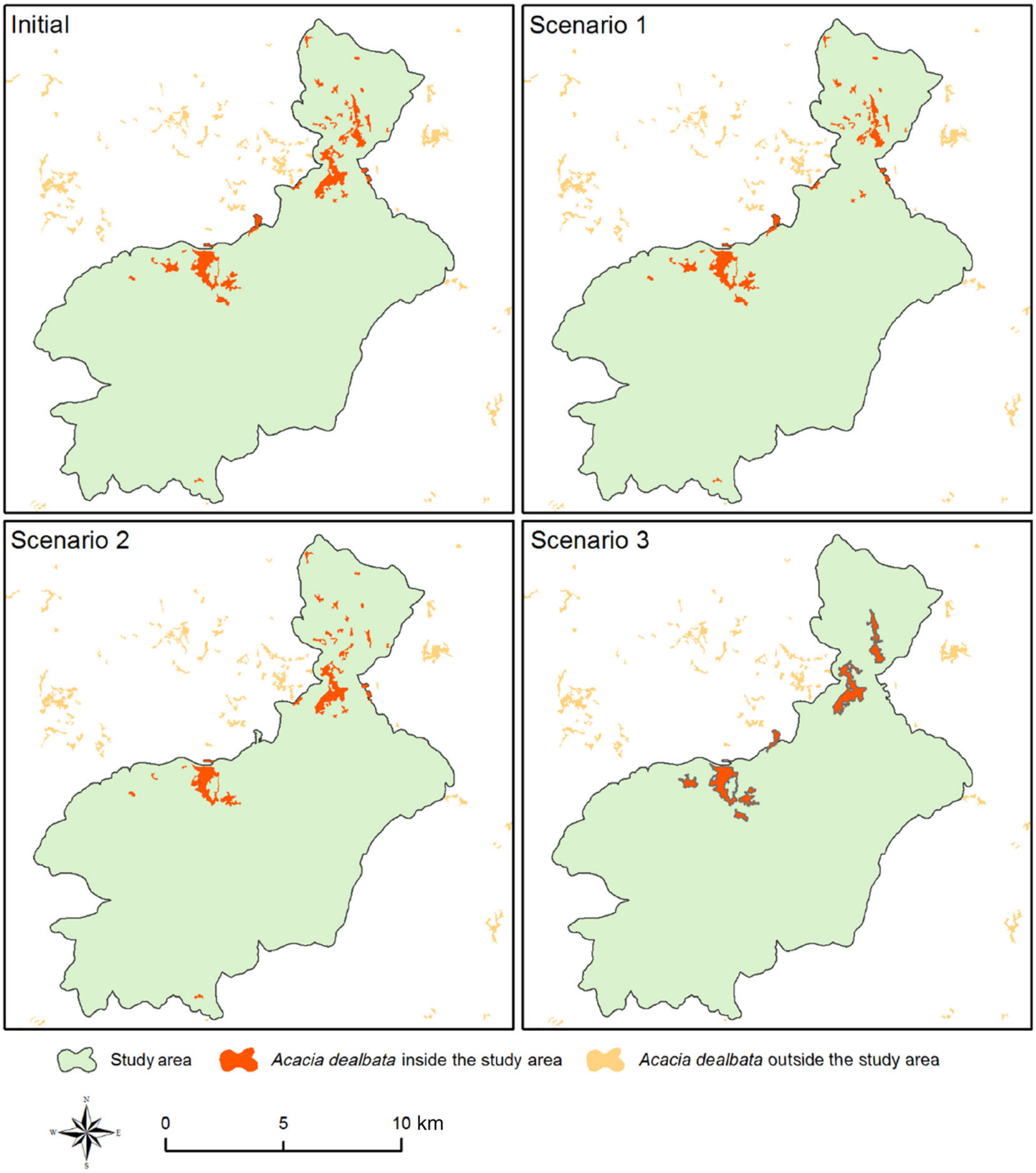


Figure 2. Initial and scenarios' maps. In all scenarios the area of *Acacia dealbata* removed is similar. In scenario 1 the largest patch is removed. In scenario 2, five patches with intermediary perimeter–area ratio values are removed. In scenario 3, the 25 patches with higher perimeter–area ratio are 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 overlooking 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)**

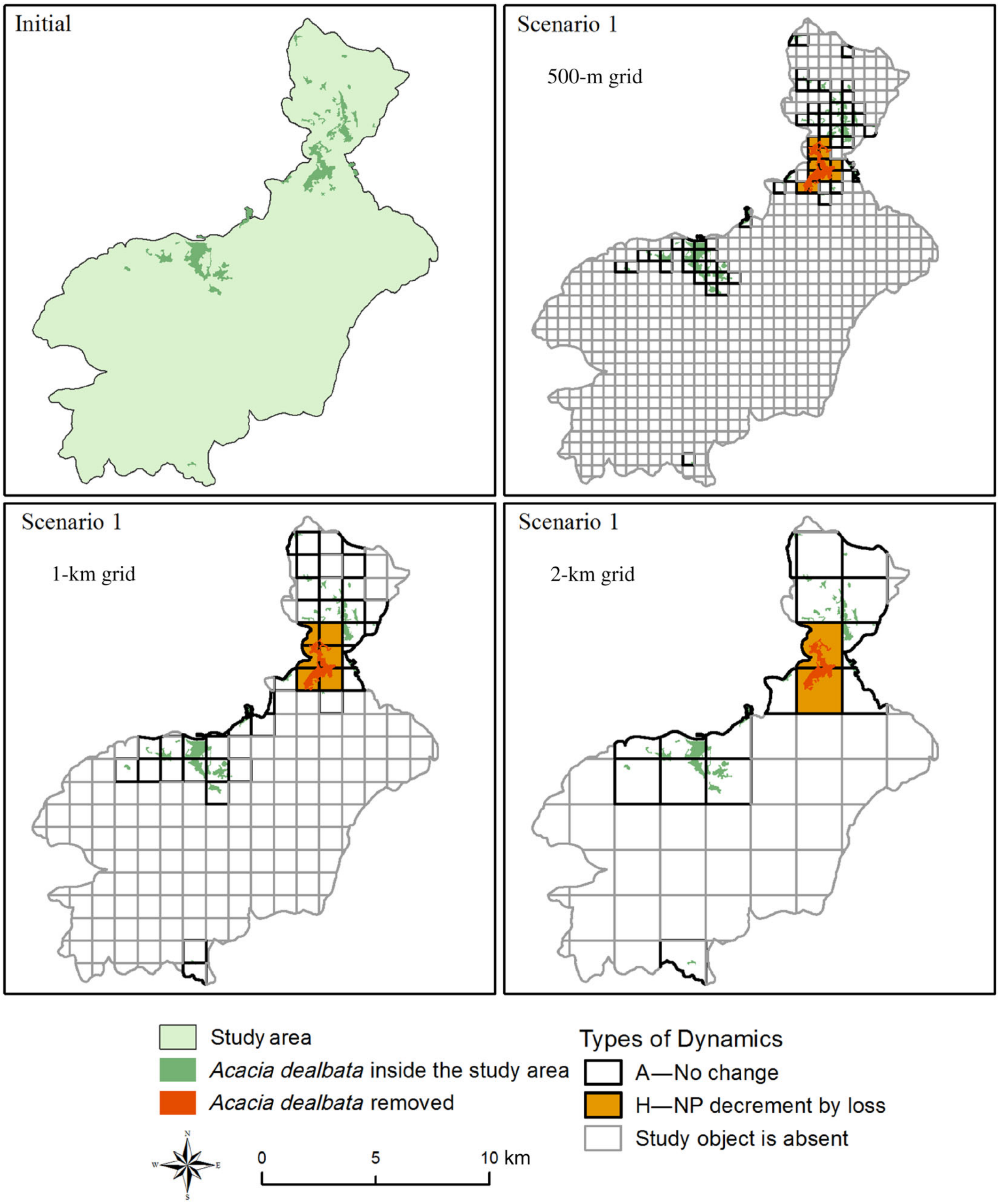


Figure 3. Types of dynamics for scenario 1 calculated using different grid sizes.

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–m^2, 1,000 × 1,000–m^2, and 2,000 × 2,000–m^2 grids.

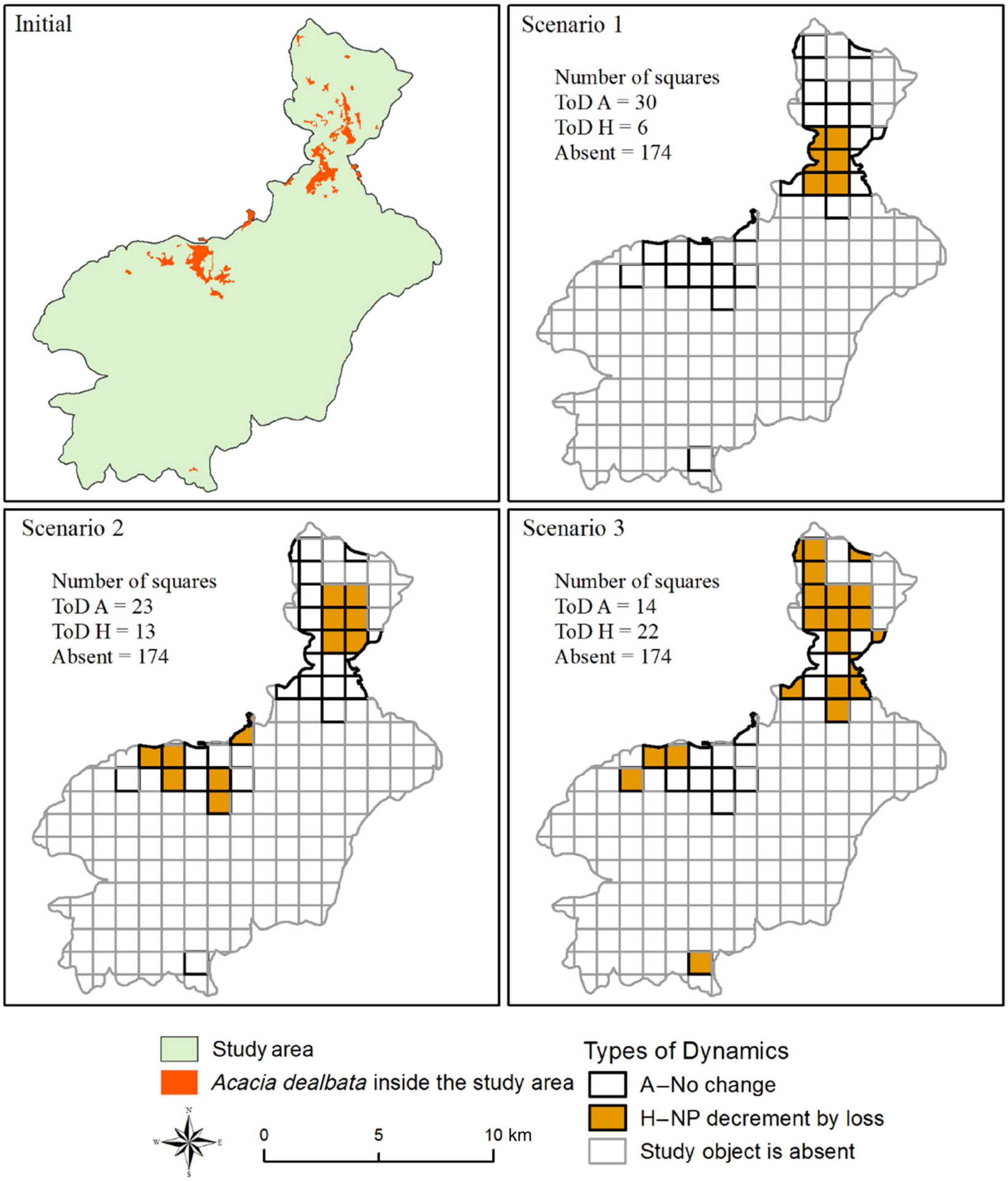


Figure 4. Types of dynamics produced by the different scenarios.

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:

$$\mathrm{d}M\,(\%) = 100.\frac{M - M_{\mathrm{after}}}{M}$$

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 ($\mathrm{IIC}_{\mathrm{num}}$) (Pascual-Hortal & Saura 2006) given by:

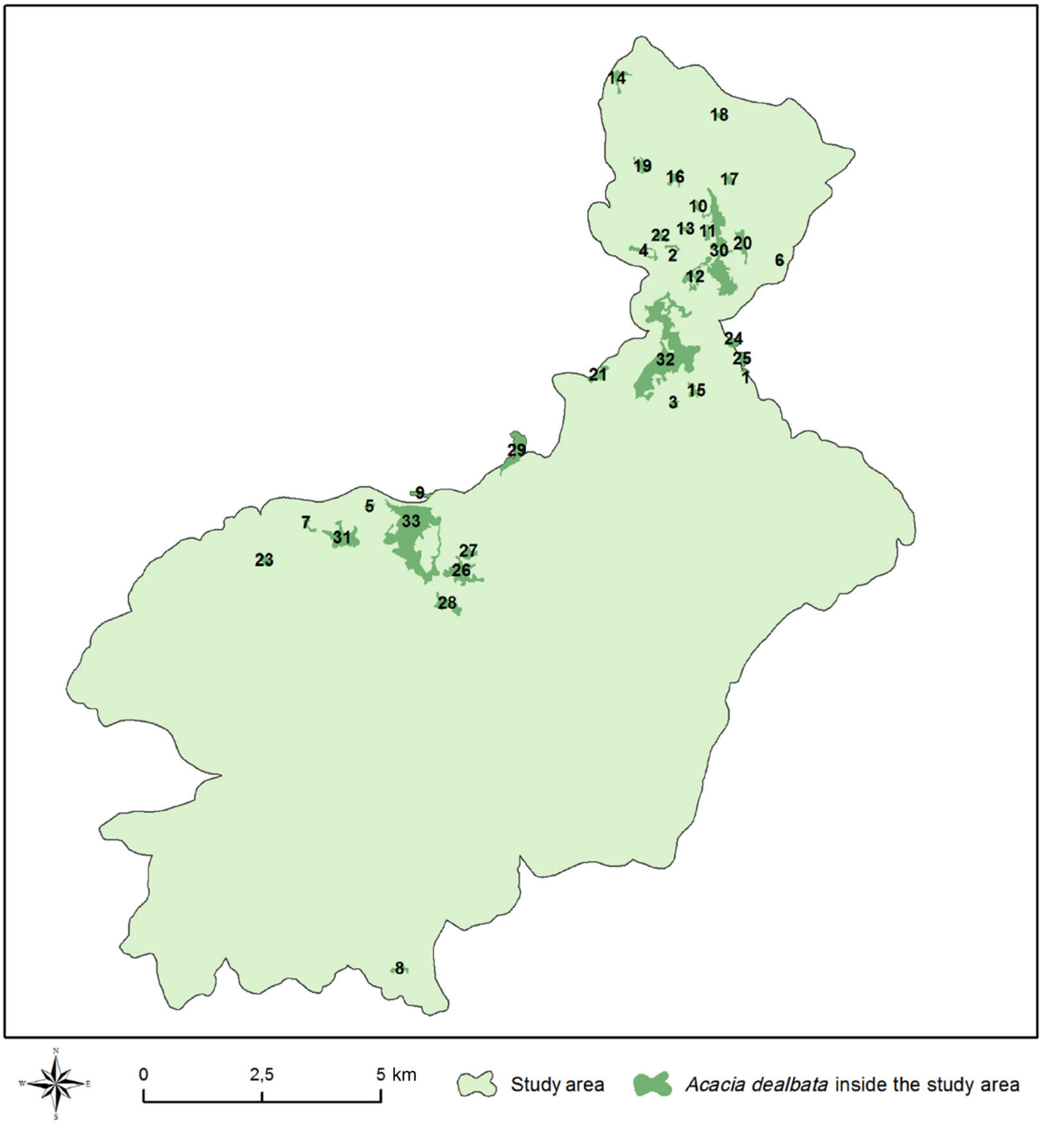


Figure 5. Ranking of *Acacia dealbata* patches based on their relative importance for the species' overall connectivity in the study area. Calculated using perimeter–area ratio as the key attribute. From 1—patch that contributes the most, to 33—patch that contributes the least.

$$\mathrm{IIC}_{\mathrm{num}} = \sum_{i=1}^{n} \sum_{j=1}^{n} \frac{a_i . a_j}{1 + nl_{ij}}$$

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.

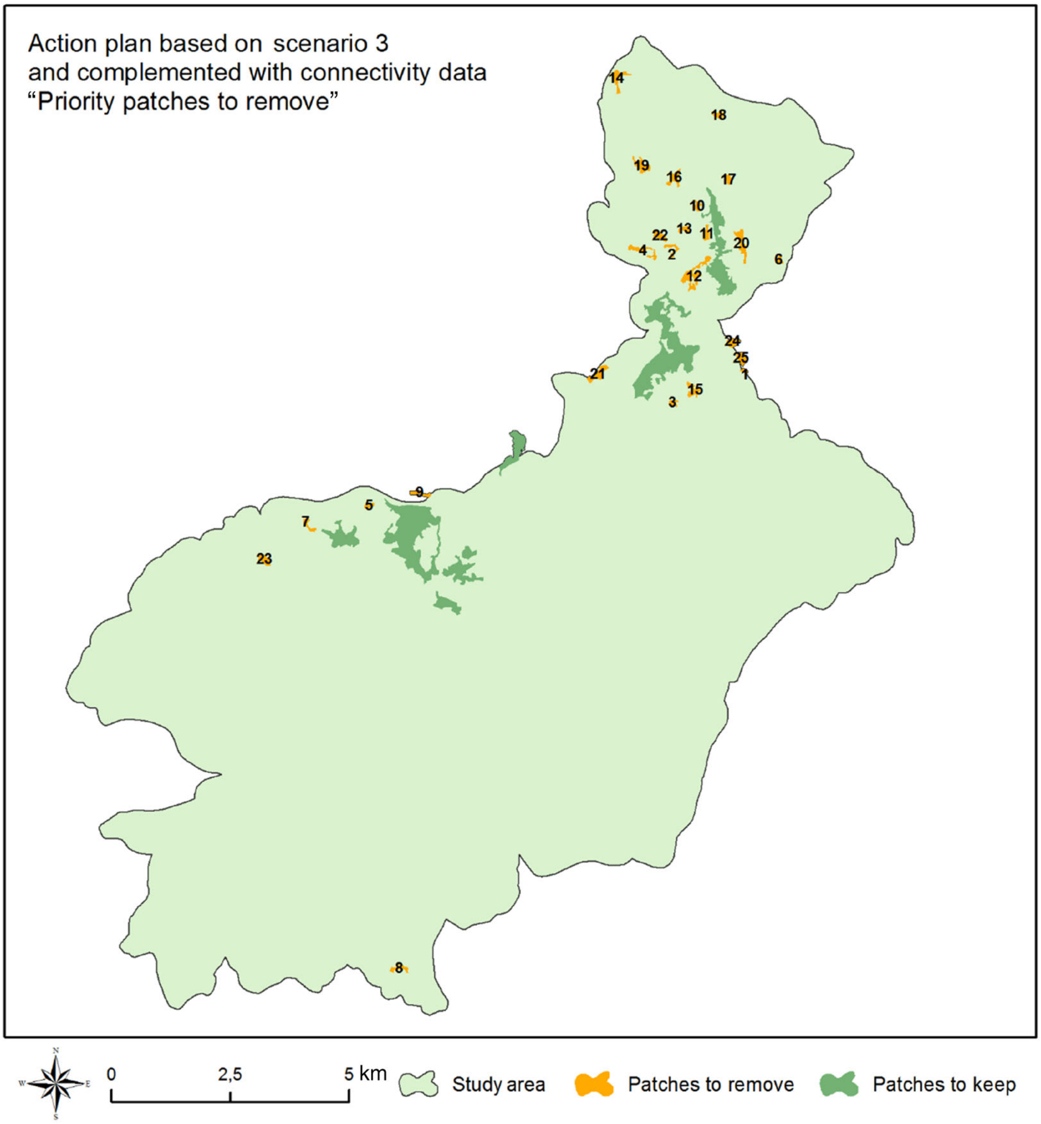


Figure 6. Ranking of *Acacia dealbata* patches to remove toward scenario 3 and considering connectivity aspects.

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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