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Forest expansion in mountain protected areas: Trends and consequences for the landscape

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HIGHLIGHTS

- The three Parks exhibited similar processes of afforestation between 1956 and 2016.
- Gains in forest result from densification and aggregation of small forest patches.
- Afforestation is related to reduction of livestock during summer in subalpine areas.
- We report a common pattern of morphological & functional landscape homogenization.

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ABSTRACT

Mountain regions in Western Europe have gone through a massive rural–urban migration and the collapse of their traditional socioeconomic system. As a result, forest has occupied many old pastures and croplands. In protected areas – such as National Parks – changes in the landscape can affect biodiversity and other services, including the values that motivated their declaration. Any policy decision in these areas requires quantifying the extent and impact of land-cover changes and their consequences on landscape structure and functioning. In this study we analyze the patterns of change in forest cover during six decades in three mountain National Parks in Spain. Our aim is to quantify those patterns, their effects on the landscape, and discuss the potential consequences for the main natural values and services. We assessed changes in forest cover through reclassification of aerial ortophotographs taken in 1956–57 (*past images*) and 2016–17 (*recent images*). The three Parks show a relatively low change in total forest area (+5–10%), and a much larger increase in dense forest (+20–30%), with an important effect of land-use legacies, and similar patterns of landscape homogenization. There were fewer but larger forest patches in 2016 than in 1956, and most of the gain in dense forest occurred in core areas (+20%), while transition areas such as edges, bridges or loops decreased between 30 and 55%. Given their potential consequences on biodiversity and other services, these patterns of land-cover change and landscape configuration should be explicitly considered when designing the sustainable management of abandoned landscapes in protected areas.

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1. Introduction

During the 20th century, most Western Europe has gone through a process known as forest transition, i.e., the shift from net loss to net gain in forest area (Barbier et al., 2010; Mather, 1992). This is one of the dominant land use change processes in Europe (van den Zanden et al., 2017), and has been the consequence of massive agricultural land abandonment since the 19th century. Farmland abandonment primarily occurred in less productive areas such as remote and mountainous regions (Kuemmerle et al., 2016; Lasanta et al., 2017), while land uses intensified in the most suitable locations to match the required increase in agricultural production (Levers et al., 2018; van den Zanden et al., 2017). In Spain, these processes occurred a few decades later than in other countries due to its historic delay in industrialization and the autarchy period that followed the Civil War (Kauppi et al., 2018; Valbuena-Carabaña et al., 2010). From the 1960′s onwards, the spectacular economic growth led to a massive rural–urban migration and to the collapse of the traditional socioeconomic system of rural regions (García-Ruiz et al., 1996; Gómez-Limón & Fernández, 1999), resulting in a dual production system of abandonment and intensification (Bernués et al., 2011; Levers et al., 2018). Many traditional primary sector activities were abandoned, particularly in marginal remote mountain areas (MacDonald et al., 2000; Morán-Ordóñez et al., 2013), where forest has occupied many old pastures and croplands (Ameztegui et al., 2010; Cervera et al., 2019).

The consequences of land abandonment on landscape vegetation dynamics and ecological processes in Europe have received considerable attention, although mostly at local scales (Lasanta et al., 2017). Land-abandonment triggers secondary succession processes with either positive or negative impacts on the habitats and species inhabiting these landscapes (Rey Benayas, 2007; van den Zanden et al., 2017): for example, the loss of agroforestry landscape mosaics to continuous forest cover can reduce overall species richness – despite forest specialist being favored – and is commonly associated with increased fire risk and reduction of river flows, particularly in the Mediterranean (Lasanta-Martínez et al., 2015; Mantero et al., 2020). On the other hand, carbon sequestration is generally enhanced, and landscapes increase their role in hydrological regulation and erosion reduction (Rey Benayas, 2007).

In National Parks, areas of great cultural and natural value, landscape changes resulting from forest expansion can significantly affect biodiversity values and ecosystem service provision (De Pablo et al., 2020; Locatelli et al., 2017; Morán-Ordóñez et al., 2013), and go against the very values that motivated their declaration as protected areas. In Spain, National Parks are granted with the maximum legal protection to nature and most uses are strictly regulated or even forbidden, which has often been a source of intense conflict with the local communities (Calvache et al., 2015; Guadilla-Sáez et al., 2020; López & Pardo, 2018). Despite the strict protection, high-mountain National Parks in Spain have been acknowledged as social-ecological systems heavily influenced by people (Cumming & Allen, 2017), where extensive livestock breeding is allowed or even encouraged (De Pablo et al., 2020). This activity has been understood as part of the set of manifestations derived from the historical presence of man in the area and considered a heritage worth preserving. Therefore, Spanish National Parks are not oblivious to the socio-economic changes of mountain areas and can also drag along legacies of changes in use prior to their declaration as a protected area. Currently, some of the habitats of conservation importance in those areas (e.g. shrub-grasslands mosaics) are semi-natural in origin and therefore dependent on the continuation of traditional management practices. Hence, any policy decision requires a good quantification of the extent and impact of land-cover changes, and an assessment of the consequences on the structure and functioning of the landscape (Martínez-Vega et al., 2017; Morán-Ordóñez et al., 2011).

In this study, we analyze the patterns of change in forest cover during the last six decades in the three mountain National Parks in northern Spain: "Picos de Europa", "Ordesa y Monte Perdido", and "Aigüestortes i Estany de Sant Maurici". We compared forest cover in images from the so-called 'American flight', which covered the whole peninsular Spain between 1956 and 1957, with the most recent images available, so that we covered the widest temporal gradient possible. Our objective was threefold: (1) to quantify the changes in forest cover that have occurred in the last 60 years in the three parks studied; (2) to assess the spatial patterns of such changes; and (3) to analyze the impacts of such changes on different landscape metrics. We finally discussed the consequences on biodiversity and conservation of forest ecosystems.

2. Materials AND METHODS

2.1. Study area

Our study area includes the three Spanish National Parks located within the Cantabro-Pyrenenan range: "Picos de Europa", "Ordesa y Monte Perdido", and "Aigüestortes i Estany de Sant Maurici" (Fig. 1). These three parks share a series of characteristics, both physical and socioeconomic, which motivated their selection. First, they are the only mountain National Parks located in Spain's northern fringe and represent similar topo-climatic conditions. Moreover, they were the first three Parks to be declared in Peninsular Spain (1918, 1918 and 1955, respectively), so they have been under protection for a sufficiently long time to appreciate the potential effects of the protection. Finally, the three parks are in remote mountain areas where livestock grazing in summer pastures has been the main traditional activity. This activity has persisted to date in the three parks, but it is not clear to what extent the parks have been subjected to the same processes of rural exodus than their surrounding areas.

Picos de Europa National Park (hereafter *Picos*) was created in 1918 and enlarged in 1995, currently covering 67,455 ha in the heart of the Cantabrian range (43°11′51″ N, 4°51′06″ W). *Picos* comprises impressive limestone massifs separated by deep gorges carved by rivers flowing to the north, with elevation ranging from 75 m to 2,648 m (Table S1.1; Jiménez-Sánchez et al., 2014). Vegetation is highly conditioned by the short distance to the Cantabric sea (~40 km), and the Park hosts some of the best examples of Atlantic mixed hardwoods in Spain, including *Quercus petraea* (Matt.) Liebl., *Castanea sativa* Mill., and *Corylus avellana* L. as some of the dominant species. *Fagus sylvatica* L. dominates most forested areas above 1,000 m, whereas vegetation above 1,500 m is commonly restricted to subalpine pastures. Opposed to the other two National Parks, Picos has maintained population – albeit small – within its limits (a total of 1157 inhabitants as in 2020), whose economy is based on the primary sector, mostly livestock breeding (López & Pardo, 2018). In the last decades, the economy has undergone a strong conversion towards the tertiary sector, with a drastic drop in sheep farming (-80% in the last 15 years), and several municipalities are at risk of being depopulated in the next few years.

Located in the Central Pyrenees, the "Ordesa and Monte Perdido" National Park (42°40'18" N; 0°3'20" E; hereafter *Ordesa*) was also created in 1918 to protect its complex relief, dominated by the largest calcareous massif in Europe and affected by intense glacial erosion. Covering originally 2,100 ha, it was enlarged in 1982 for a total of 15,608 ha (plus 19,679 of the buffer zone, also protected but with fewer restrictions), in which elevation ranges from 700 to 3,355 m (Table S1.1). The relief, which includes 34 peaks exceeding 3,000 m, dramatic vertical cliffs, deep canyons and U-shaped valleys (García-Ruiz et al., 2014), exerts a large effect on vegetation. *Ordesa* hosts almost 1400 vascular plants – one fifth of the total of the Iberian Peninsula – 180 species of vertebrates and 600 arthropods (García-Ruiz et al., 2014). Up to 1,500–1,700 m, there are extensive forests of *Fagus sylvatica*, *Abies alba* Mill., and *Pinus sylvestris* L., whereas at higher elevations up to 2,000 m, *Pinus uncinata* Ram. ex DC. dominates. Above this elevation, we find subalpine grasslands, which in fact constitute three quarters of the total surface of the Park. These subalpine grasslands have been used for millennia as summer pastures for livestock (Fillat et al., 2008),

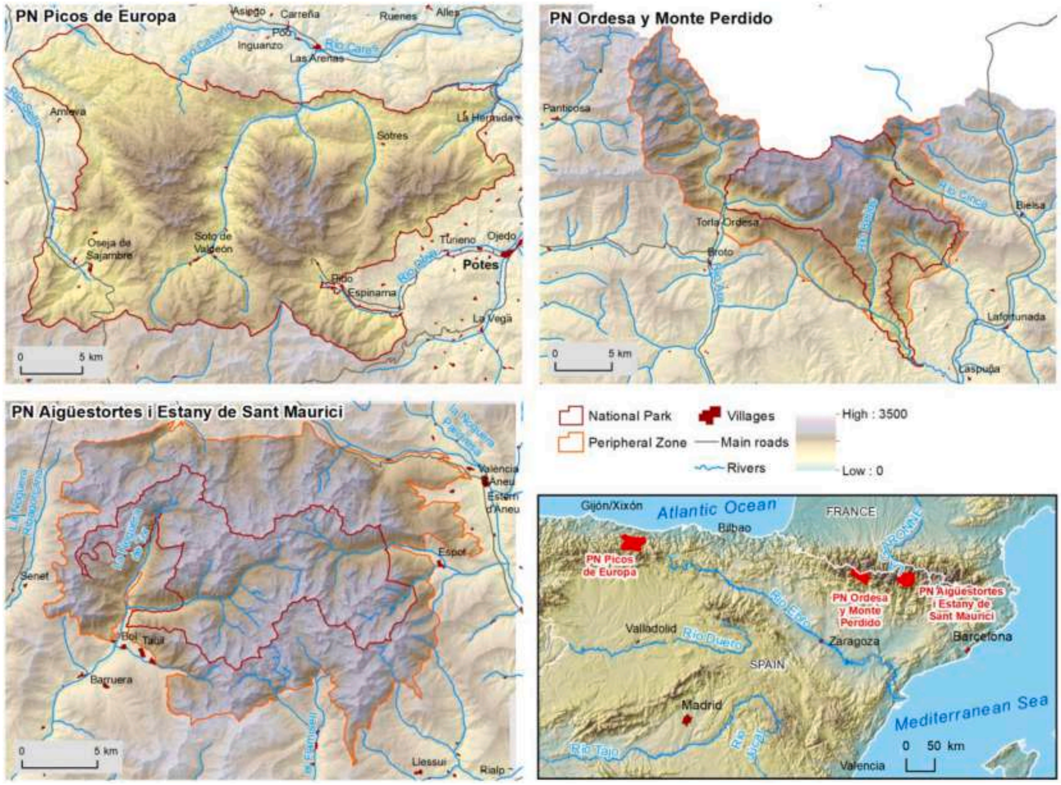

Fig. 1. Location of the study area showing the position of the three National Parks in Spain (bottom right), and the limits and relief for Picos de Europa, Ordesa y Monte Perdido, and Aigüestortes i Estany de Sant Maurici National Parks.

particularly sheep herds which often traveled hundreds of kilometers from the lowlands of the Ebro valley. Stock density has fallen considerably after its peak at the beginning of the 20th century (Fillat et al. 2008). The sharp decline in grazing is related to the fact that the Park is located in a sparsely populated and heavily aged region, as well as the low economic profitability of extensive cattle ranching. The development of a strong economic activity related to tourism has haltered depopulation trends, but has also led to problems in farm generational turnover and to further farm abandonment (Muñoz-Ulecia et al., 2021).

The "Aigüestortes i Estany de Sant Maurici" National Park (hereafter *Aigüestortes*) was created in 1955 and currently covers 14,119 ha (plus 26,733 ha of the buffer zone) in the high mountain domain of the Pyrenees massif, and elevation ranges from 1,300 m to greater than 3,000 m at the highest peaks (Table S1.1). The main valleys in *Aigüestortes* run from west to east, defining two clearly contrasted slopes (dry and wet) with very different conditions for vegetation (Gracia et al., 2011; Villero et al., 2015). The vegetation is mostly subalpine, and *Pinus uncinata* and *Abies alba* occupy most of the area, with some presence of *Pinus sylvestris* at the lower valleys. Above 2,300 m, vegetation is composed of subalpine grasslands. Until the establishment of the National Park, the traditional economy of the villages in the region was based on agriculture, cattle-raising and forest management (Abella, 2002). Due to the complex relief, cultivable lands represented a small portion of the territory, while production of pastures and forage for livestock (mainly sheep, cows and horses), represented one of the most important land uses (Gracia et al., 2011). Since its creation in 1955, the exploitation of natural resources (except for traditional livestock raising) has been strictly prohibited within the National Park, although certain uses (logging, etc.) have been allowed in the buffer zone (Abella, 2002; Villero et al., 2015).

2.2. Changes in forest cover (1956–2016)

We assessed changes in forest cover for the three National Parks through reclassification of aerial ortophotographs taken in 1956–57 (*past images*) and 2016–17 (*recent images*). The *past images* are part of the so-called "American flight", the first photogrammetric flight in Spain, which consists on a set of aerial images taken by the United States Army Map Service, in collaboration with the Spanish Army's Geographical Service (SGE) and the National Geographic Institute (IGN). They cover the entire national territory in black-and-white at a scale of 1:33.000 and a resolution of 1 m. These images were georreferenced and orthorectified by the IGN. The *recent images* were taken within the National Plan of Aerial Orthophotography (PNOA) of the IGN, and are RGB images at a resolution of 25 cm.

Each *past* and *recent* image was semi-automatically reclassified using the Supervised Classification tool in ArcMap 10.7 (ESRI Inc., USA), into a binary raster with 'tree' and 'non-tree' values with a resolution of 1 m, and we stitched all the reclassified images into a *past* and a *recent* mosaic per each Park. We then created a 0.25 ha sampling grid covering each Park and buffer zone. For each cell of the sampling grid, we determined past and recent forest canopy cover (FCC) as the ratio between the number of 'tree' pixels and the total number of pixels in the cell, expressed as a percentage (Ameztegui et al., 2010). We calculated the net change in FCC for each cell as the difference between the *recent* and *past* FCC.

For each time moment (past and recent) we classified each cell in four cover classes according to their FCC: (i) *non-forested*, when FCC $<$ 10%; (ii) *sparse forest*, when 10% $\leq$ FCC $<$ 20%; (iii) *open forest*; when 20% $\leq$ FCC $<$ 40%; and *dense forest*, when FCC $\geq$ 40%. We then constructed a transition matrix to determine the changes between the different FCC classes in each National Park, and we computed several transition metrics for each forest cover class to better characterize their dynamics: gain, loss, total change, swap and net change (Pontius et al., 2004). Gains for a given land cover i can be defined as those areas that did not belong to i at time 1 but have converted to i at time 2. Conversely, losses are those patches that have converted from i in time 1 to any other land cover in time 2. Total change is the sum of gains and losses for a given land cover, while the net change is the difference between gains and losses. Finally, swap represents a given quantity of *i* loss at one location that is accompanied by the same quantity of *i* gain at another location – so no net change has occurred – and can be calculated as two times the minimum of the gain and loss (see Pontius et al., 2004). We also determined the changes in forest cover separately for the area included within the boundaries of each park and for their corresponding buffer zones (except in *Picos*, which has no buffer zone).

2.3. Landscape metrics

For each National Park and study date (1956 and 2016) we calculated several landscape ecology metrics using Fragstats v4.2 (McGarigal et al., 2012): number of patches, mean patch size, edge density, perimeter to area ratio, mean shape index and Shannon diversity index, calculated as:

$$SDI = -\sum_{i=1}^{R} p_i \ln p_i$$

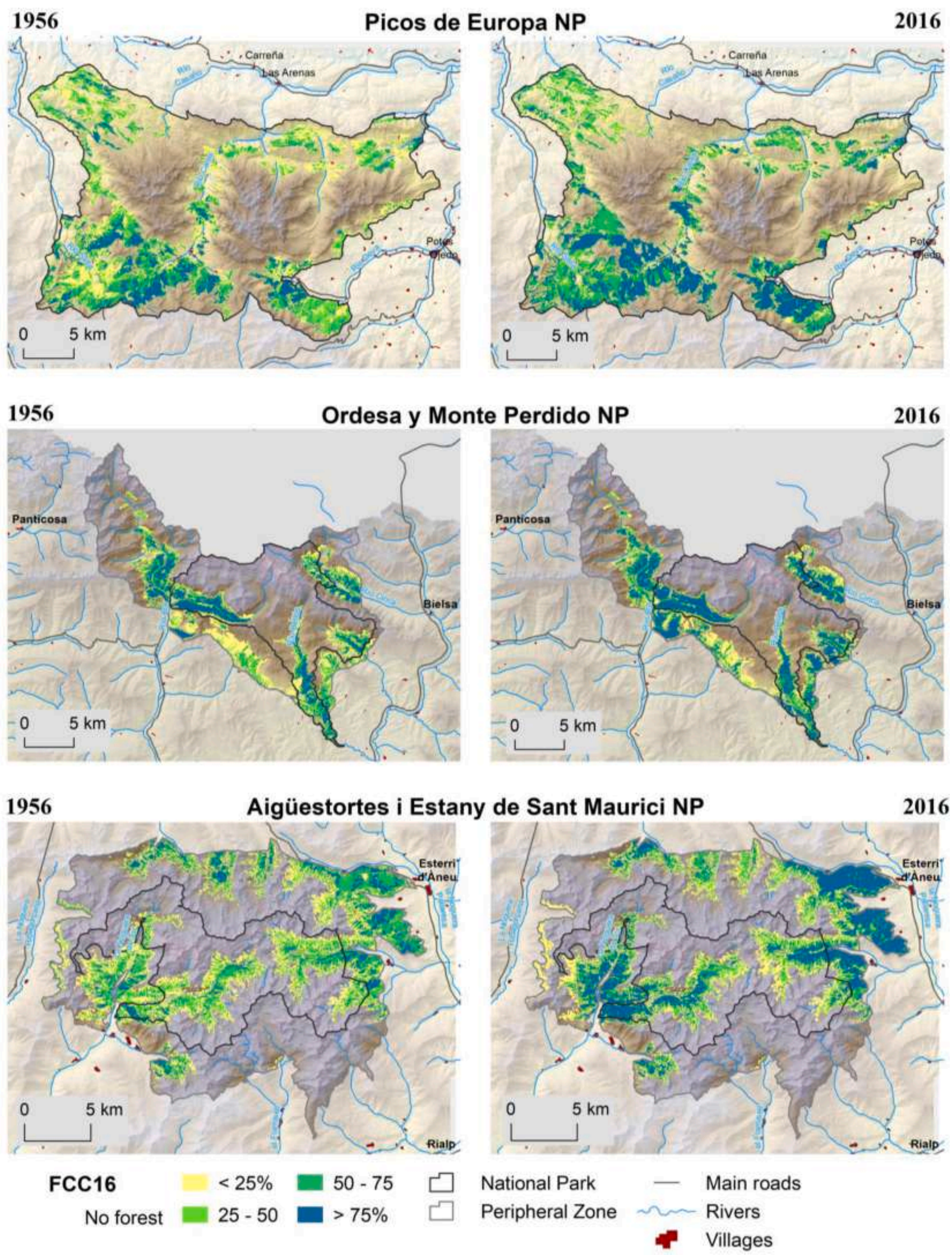

Fig. 2. Forest canopy cover in 1956 and 2016 for the three studied National Parks. Forest cover is determined as percentage of forest cover in 0.25 ha cells based on the photointerpretation of aerial ortophotographs. ESRI Map Packages (.mpk) including the map document and the layers can be downloaded as Electronic Supplementary Material.

where p_i is the proportion of the landscape occupied by the forest cover class i. This index increases as the number of different patch types increases and/or the proportional distribution of area among patch types becomes more equitable. The description of all the other landscape metrics can be found in Supplementary Material S4.

We also performed a Morphological Spatial Pattern Analysis (MSPA) using Guidos Toolbox (Vogt & Riitters, 2017). Fragstats allows for a quantitative analysis of landscape patterns by computing a wide variety of standard landscape metrics at the landscape or class level, whereas MSPA is a pixel-based landscape classification technique targeted at the description of the geometry and connectivity of the image components (Soille & Vogt, 2009; Vogt et al., 2007). MSPA segments a raster forest binary map (i.e. forest vs. non-forest) into seven mutually exclusive classes: core, islet, connectors (bridge and loop), boundaries (edge and perforation), and branch (Soille & Vogt, 2009) (See Supplementary Material S2 for a description of the MSPA classes). The MSPA emphasizes the geometric arrangement of these elements and particularly the connectivity among them, so it complements approaches based on standard landscape metrics such as Fragstats (Martínez-Vega et al., 2017; Saura et al., 2011).

3. Results

3.1. Changes in forest cover

The three National Parks have a similar proportion of the territory covered by forest (around 30%). For each Park, the amount of forested area is also very similar within the borders of the National Park and in their corresponding buffer zones, and always ranges between 25 and 30% (Fig. 2; Table S3.1). The three Parks also show a similar change in total forest area between 1956 and 2016, ranging from an increase of 5.2% in *Aigüestortes* to 10.3% in *Picos* and 11.5% in *Ordesa* (Table 1). In all of them, the increase in forest area was mainly due to a very relevant increase in the surface of dense forest (always greater than 20%), while the surface of open or sparse forest decreased considerably, particularly in *Picos* (Table 1).

While most of the patches of dense forest in 1956 have remained unchanged in the studied period, more than half of the sparse and open forest in 1956 had become dense forest in 2016 (Fig. 3). This trend was particularly important in *Picos* and *Ordesa*, and showed more inertia in *Aigüestortes*, where up to 38% of the sparse and open forest remained in the same state 60 years later (Fig. 3). In *Picos* and *Ordesa*, more than 80% of the surface classified as sparse forest in 1956 had changed into either open forest (27 and 26%, respectively) or into dense forest (57 and 53%) by 2016.

Accordingly, dense forest was the category with highest surface gain in the three parks, although not necessarily the most dynamic one. Open forest also underwent high rates of total change, but due to the high proportion of both gains and losses, the amount of net change was always considerably low, as indicated by the high proportion of swaps between categories (Table S3.3).

Table 1
Total area by forest cover class and Park, and relative changes between 1956 and 2016.Percentage of change represents the difference in total surface of each forest cover class, as compared to its surface in 1956.

	Area 1956 (ha)	Area 2016 (ha)	Change (%)
Picos de Europa NP			
Non forested	45,106.4	43,206.5	-4.2
Sparse	2,082.7	1,176.2	-43.5
Open	3,975.3	2,602.6	-34.5
Dense	12,439.1	16,618.1	33.6
Total forest	18,497.1	20,396.9	10.3
Ordesa y Monte Perdido NP			
Non forested	26,082.4	25,076.2	-3.9
Sparse	1,007.7	739.1	-26.7
Open	1,668.2	1,339.4	-19.7
Dense	6,076.8	7,680.4	26.4
Total forest	8,752.7	9,758.9	11.5
Aigüestortes i Estany de Sant Maurici NP			
Non forested	28,689.9	28,101.2	-2.1
Sparse	1,628.2	1,287.9	-20.9
Open	2,779.5	2,176.2	-21.7
Dense	6,936.2	8,468.6	22.1
Total forest	11,343.9	11,932.6	5.2

3.2. Landscape metrics and morphological spatial pattern Analysis

The abovementioned patterns of afforestation had consequences on the landscape of the three National Parks. In all cases, the overall total number of forest patches, the edge density and the diversity of patches decreased, whereas mean patch size increased in ca. 15% (Table 2). The patterns were nevertheless very different for the different forest cover classes: whereas both the number of patches and their size decreased for sparse and open forests, in the case of dense forest the mean patch size almost doubled between 1956 and 2016. The increase was particularly noticeable in *Picos*, where the mean dense forest patch size increased from 18 to 50 ha (Table S4.1). Consequently, the landscape homogenized, with less open areas, zones of transition (edges) and lower diversity of patches (Table S4.1).

The Morphological Spatial Pattern Analysis also revealed an important homogenization of the landscape. In 1956, most of the forest area of the three parks was present as relatively large patches of forest (*cores*): this category represented 55% of the total forest surface in *Aigüestortes*, and up to 65% in *Picos* and *Ordesa*. An important part of the forest surface (between 20 and 30%) was found at the edges – transition zones between the cores and the background matrix – whereas the rest of the classes were much less represented (Fig. 4 and Figures S2.2 – S2.4). Between 1956 and 2016, the proportion of *core* areas increased by a very similar proportion in all three parks: +19.5% in Aigüestortes, +20.7% in Picos and + 21.9% in Ordesa (Fig. 5), whereas most of the other classes diminished their presence. The biggest drop was that of the *bridges* – linear structures that connect two or more core areas – which were reduced by 30 to 55%, and borders (*edges* and *perforations*), from which almost 50% had converted into core areas by 2016 (Fig. 5). In the case of *islets* – isolated forest patches that are too small to contain core pixels – Ordesa and Picos experienced a marked decrease (- 47 and −88%) while in Aigüestortes *islet* areas increased by 15%.

The transition matrices between MSPA elements showed two different patterns in the dynamics between classes. In *Picos* and *Ordesa* most of the linear elements in 1956 (*bridges*, *branches* or *loops*) had become *core* or *edge* areas as a result of their absorption into larger patches of forest, whereas only a quarter remained in the same MSPA class. In these two parks, in addition, the increase in *core* areas was produced by contributions from all classes, including a substantial part of the *islets*, which in 2016 appear mostly connected with other forest areas, either as *core*, *edge* or *branches.* On the other hand, in *Aigüestortes* the linear elements and the *islets* showed much more inertia, and the increase in the core area was mainly due to the incorporation of *edge* zones and the disappearance of openings in the continuous forest patches (i.e. *perforations*).

4. Discussion

4.1. Changes in forest cover in mountain National Parks (1956–2016)

The increase in total forest cover observed in the three studied National Parks ranged between 5 and 11%, a lower increase than that reported for the same period in similar non-protected mountain areas, which usually approaches 15–20% (Álvarez-Martínez et al., 2014; Ameztegui et al., 2010), but higher than overall forest expansion values reported for Natura 2000 sites at the European level (net change around 1%; (Hermoso et al., 2018)). The low rate of forest expansion in mountain National Parks comes as no surprise (see for example De Pablo

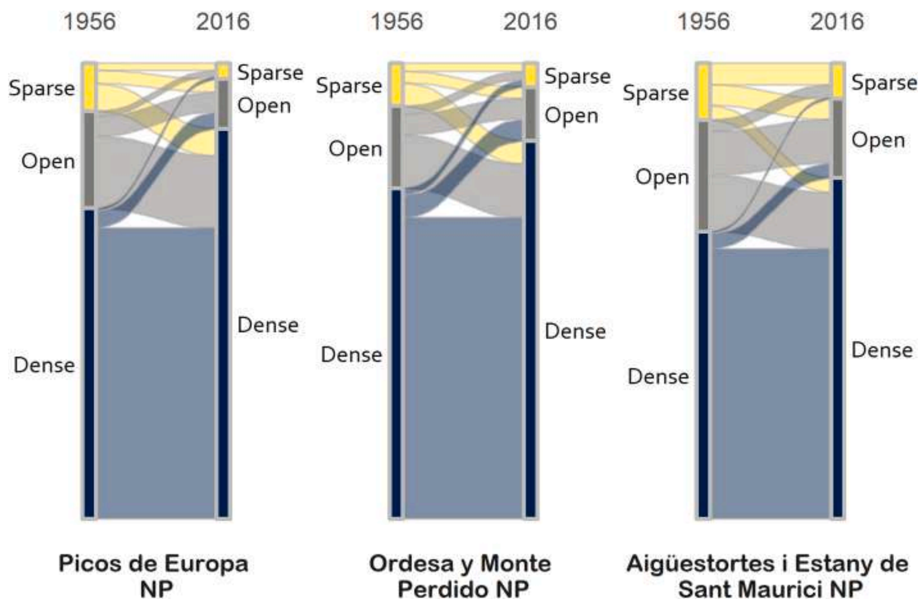

Fig. 3. Alluvial plot showing the transitions between the three categories of forest canopy cover between 1956 and 2016 in the three studied National Parks. Most changes between the two dates come from "densification" of the forest, i.e. transitions from "open" and "sparse" forest into dense forests. Transitions from and to the "non-forested" category are excluded for clarity. The transition matrix for each National Park can be found in Table S3.1.

Table 2
Landscape ecology metrics for general characterization of each National Park.

Year	NP	MPS	ED	MPE	PAR	SDI
Picos de Europa NP						
1956	7,662	2.43	9.72	0.236	699.45	0.846
2016	7,076	2.90	6.85	0.199	717.84	0.601
Ordesa y Monte Perdido NP						
1956	3,812	2.33	9.38	0.219	701.96	0.818
2016	3,577	2.77	7.38	0.205	712.61	0.658
Aigüestortes i Estany de Sant Maurici NP						
1956	6,796	1.69	11.61	0.196	701.96	0.925
2016	5,974	2.02	9.03	0.182	710.09	0.799

NP: number of patches; *MPS*: mean patch size (ha); *ED*: Edge density (km/ha); *MPE*: mean patch edge (km); *PAR*: perimeter to area ratio (km/ha); *SDI*: Shannon's diversity index

et al., 2020; Gracia et al., 2011; Ninot et al., 2011), as protected areas typically show higher persistence in land-cover than unprotected areas (Hermoso et al., 2018; Martínez-Fernández et al., 2015), and mountain ecosystems also tend to show higher inertia than lowlands (Garcia et al., 2019). Important limitations to pine recruitment have been observed elsewhere in Spain, associated mostly to forest succession, but also to climatic constraints (Carnicer et al., 2014). Our study focused on high mountain areas, where climatic conditions are often limiting the development of tree vegetation, particularly at high elevations (Körner, 2012). A harsh climate can limit forest expansion even in areas where anthropic pressure has decreased (Batllori & Gutiérrez, 2008; Camarero & Gutiérrez, 2007). Moreover, decades or centuries of deforestation, grazing and burns have degraded many soils to the point where forest recovery may be unfeasible (García-Ruiz et al., 2020). As a consequence, the most common change after the abandonment of pastures is their successional transition to scrublands, which present a greater capacity for colonization of open spaces (Gartzia et al., 2014; Gelabert et al., 2021).

The increase in the surface of dense forest was in turn much larger – between 20 and 30% – and does correspond with previous observations in Spain outside protected areas (Ameztegui et al., 2010; Poyatos et al., 2003). The high densification rate suggests the process of forest expansion probably started before the studied period, and thus we are currently observing a late stage of the colonization process (Abella, 2002; Lasanta et al., 2017). On the other hand, the existence of numerous open forests in the 1950s points to an external factor that maintained this structure until the mid-20th century – probably grazing and/or fire use, but also the use of wood for fuel and timber (García-Ruiz et al., 2014b; Morán-Ordóñez et al., 2013) – and that, after disappearing, led to the closure of the forest cover.

The extent of the densification process was similar in the original park surface than in the subsequent enlargements (data not shown), as reported by previous studies (Alados et al., 2011; Garcia et al., 2019; Gracia et al., 2011). This trend suggests that the land cover dynamics of these areas do not mainly depend on its conservation regime but on the historical socio-economic development of the area (De Pablo et al., 2020). Livestock breeding – associated with transhumance – has shaped the landscape of these Parks to the point that it was considered as part of a cultural heritage worth preserving, and was included in the Natural Resources Management plan of the three parks (Fillat et al., 2008; Gartzia et al., 2016a ;Gartzia et al., 2016b). In the last decades, the rural–urban migration and the disappearance of the traditional sheep transhumance have led to rates of farmland abandonment of between 40 and 90% and the replacement of sheep by beef cattle (Lasanta et al., 2017; Muñoz-Ulecia et al., 2021), with reductions in the number of sheep of more than 80% (Aldezabal, 2001; Benito, 2012; García-Ruiz et al., 1996).

This process affected earlier the Pyrenean parks (*Ordesa* and *Aigüestortes*), which are located in remote areas with very low population density and difficult accessibility, more poorly connected to the main cities (García-Ruiz et al., 1996; Rodríguez-Rodríguez & Martínez-Vega, 2018). In *Ordesa*, for example, the rural exodus began in the 1920′s and was accompanied by a rapid reduction in the grazing pressure, dropping to 20% by the end of the century (Alados et al., 2011; Aldezabal, 2001), and similar numbers have been reported for *Aigüestortes*. In the mountains surrounding *Picos*, which are more accessible and closer to important industrial and mining areas, significant pressure was maintained on the territory until the 1960 s-70 s (Morán-Ordóñez et al., 2011, 2013; Rescia et al., 2008). García Dory (1977) reported mining exploitations, harvests and hydraulic works occurring in the 1970 s even within the limits of the original *Picos* Park, and an increase in the number of domestic grazing animals – particularly horses – between 1960 and 1974. Rescia et al. (2008) also reported an increase in heads of livestock cattle in the northern parts of Picos from the 1990s to

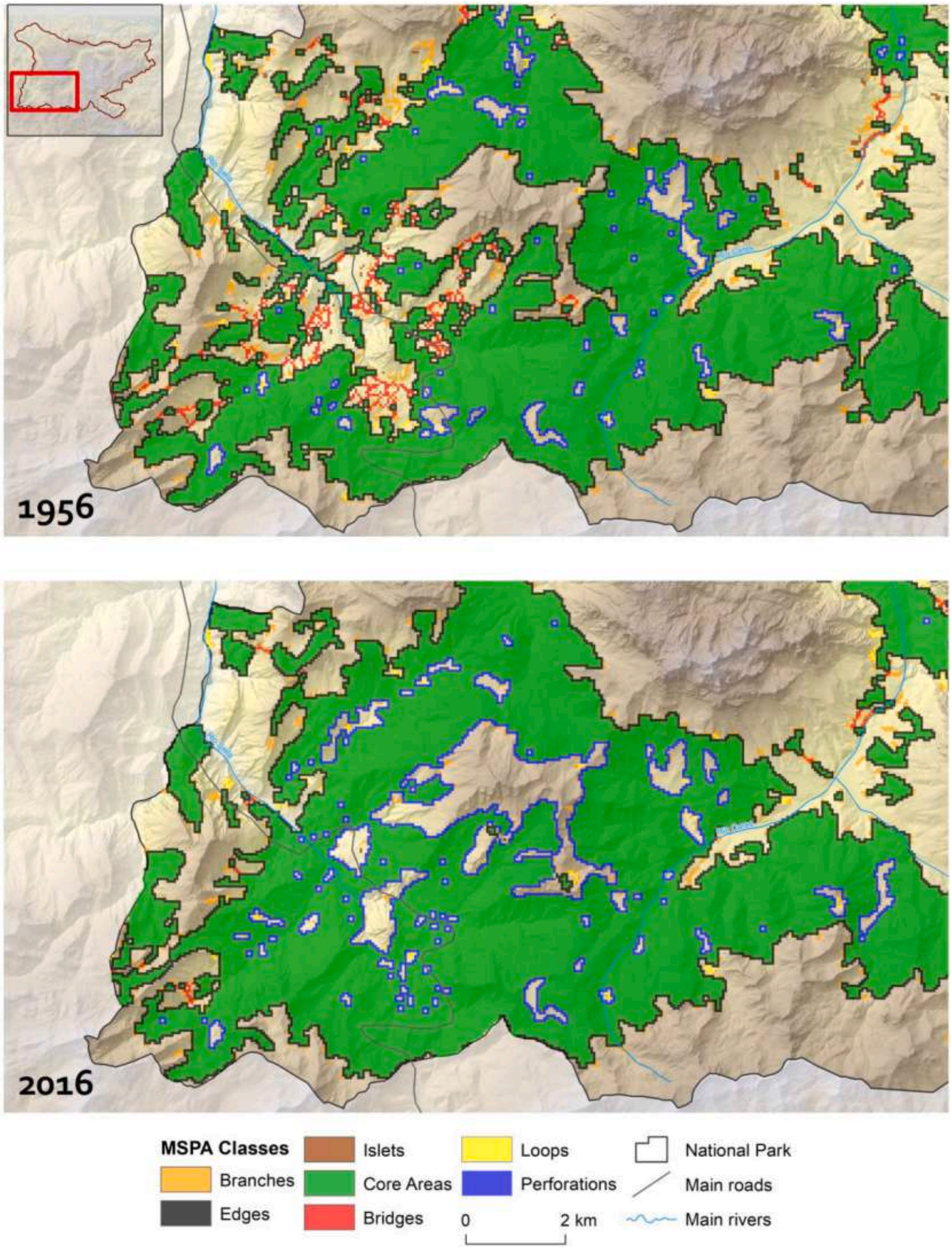

Fig. 4. Example of changes in the landscape in a portion of the *Picos de Europa* NP between 1956 and 2016 according to the Morphological Spatial Pattern Analysis. The main changes include the conversion of linear elements (*bridges*, *branches* and *loops*) into core areas. The increase in size of core areas also entails a reduction of border elements (i.e. *edges* and *perforations*). A complete view of the changes in the landscape of each park are available in Supplementary Materials S2, and can also be downloaded as ESRI Map Packages (.mpk).

the detriment of smaller livestock species – goats and sheep – which played an important role in slowing down secondary succession processes towards forest. In this regard, the higher densification rates observed for *Picos* – almost twofold those of the Pyrenean parks – could be related to the fact that land-use changes have been more recent, and the densification process is still taking place more actively.

4.2. Impacts of the functional homogenization of the landscape

The changes in forest cover led to a common pattern of landscape homogenization in all three National Parks. There were fewer but larger forest patches in 2016 than in 1956, a trend that matches previous research in mountain protected areas, including analyses of aerial photographs (Alados et al., 2011; Garcia et al., 2019; Gracia et al., 2011) and remote sensing images (Gartzia et al., 2016a ;Gartzia et al., 2016b; Martínez-Vega et al., 2017). As a consequence, both the total amount of edge habitats (i.e. ecotones) and the relative area/edge ratio decreased between 20 and 30%. The morphological spatial pattern analysis revealed that the changes in the landscape were not just morphological, but also functional. Most of the gain in dense forest occurred in core

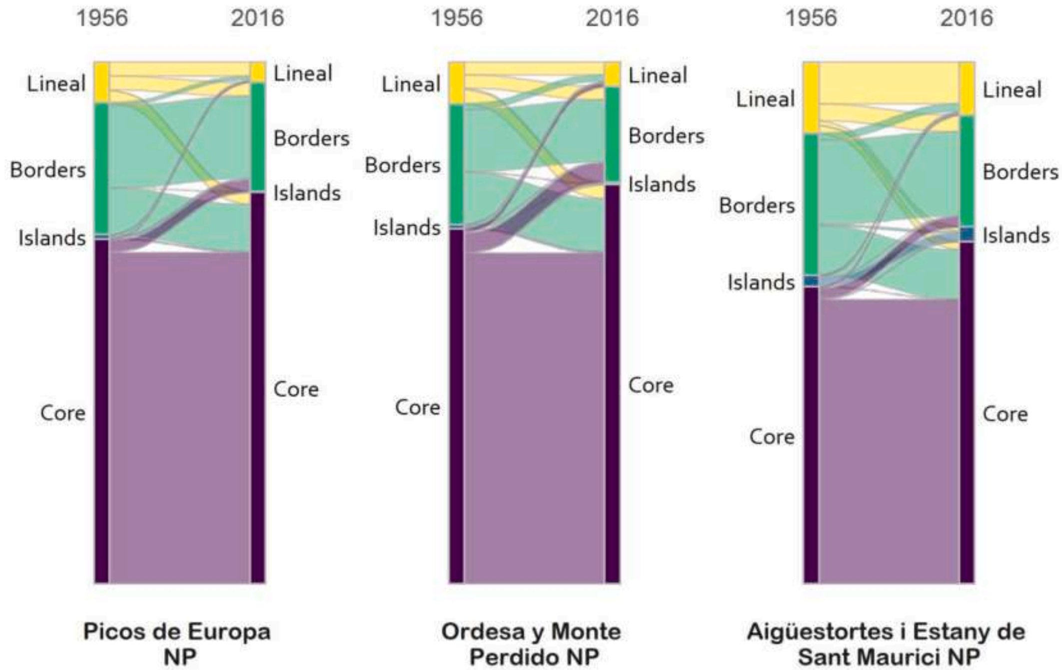

Fig. 5. Alluvial plot showing the transitions between the main structural elements according to the Morphological Spatial Pattern Analysis. The 7 categories of the MSPA are grouped into four classes for clarity. Most *linear* elements (*bridges*, *branches* and *loops*) are absorbed by the increase of *core* areas, and become either *borders* (*edges* and *perforations*) or *core* areas. As a consequence, almost a half of border elements in 1956 become core areas by 2016. The transition matrix for each National Park including the original 7 categories can be found in Table S2.1.

areas, while transition areas such as edges, bridges or loops are becoming less frequent in the three Parks. *Cores* and *bridges* are the two key pattern classes provided by MSPA in terms of connectivity (Saura et al., 2011). However, we observed that the loss of linear elements (*bridges*, *loops* and *branches*) actually occurred as they became integrated into *core* patches of forest, while the number of full isolated elements (*islets*) decreased. Moreover, we observed an unexpected inertia in linear elements and islets for *Aigüestortes,* which did not affect the total densification rates, and may be related to its peculiar landscape features, which include more than 150 lakes scattered across the landscape, as well as several large areas of bare rock. These features represent elements of vegetation discontinuity (forest patch sizes are in *Aigüestortes* almost 50% of those in the other two parks) and could have prevented further forest homogenization patterns. However, this extreme has yet to be confirmed in future research.

The most obvious effect of the advance of woody vegetation is a loss in the heterogeneity, the connectivity and diversity of grassland communities. The increase in continuous forest area necessarily entails a decrease in other habitats, especially pastures and grasslands. Both are semi-natural habitats of Community Interest (as declared in the EU Habitats Directive 92/43/EEC) which often sustain the highest levels of biodiversity in mountain areas (Alados et al., 2011; Gartzia et al., 2014; Gartzia et al., 2016a ;Gartzia et al., 2016b). These are two sides of the same coin, so if our study had explicitly focused in grasslands, we would probably be observing loss of connectivity, as reported for the Pyrenees (Gartzia et al., 2016a ;Gartzia et al., 2016b). The progressive simplification of the landscape – also observed in other European mountain systems (Campagnaro et al., 2017) – limits the development of species characteristic of transition zones and/or open areas such as mesic grasses (MacDonald et al., 2000). For instance, the increase in scrub has been reported as negative from the cultural and naturalistic points of view as it indicates a loss of traditional livestock farming practices and the decrease in meadows with high floristic richness (De Pablo et al., 2020).

However, understory plant species – some of which find in the Iberian Peninsula their southern limits of development (e.g. *Corallorhiza trifida*, *Cypripedium calceolus*, *Epipogium aphyllum*…) – may be favored by forest expansion (Garcia et al., 2019). In a recent study, García et al. (2019) observed no negative effects of land abandonment on the Pyrenean endemics or threatened plant species. As for the wildlife, the impacts associated with the expansion and densification of forests can also be variable. In areas with a large proportion of forest cover, such as the Parks studied here, we can expect substantial improvement in species typical from mature forested environments, such as the black woodpecker (*Dryocopus martius* L.) (Villero et al., 2015). On the contrary, the increase in forest connectivity implies the loss of suitable habitat for species of field margins, open areas or sparse forests, some of them of Community Interest (Birds Directive 2009/147/CE) and high conservation value such as the grey partridge (*Perdix perdix* L.) or the bluethroat (*Luscinia svecica*) (García et al., 2020).

The increase in forest area implies an increase in accumulated biomass and carbon, playing an important role in climate regulation (Varela et al., 2020). Forest spread can also modify the hydrological cycle, reducing water yield (López-Moreno et al., 2011; Morán-Tejeda et al., 2015). Such reductions are more important in the headwaters of Mediterranean rivers (such as *Ordesa* and *Aigüestortes*), where there is more irregularity in rainfall and stream flow. In this sense, Gallart and Llorens (2004) observed reductions in the flow of the Ebro River – the largest in the Iberian Peninsula – associated with increases in forest cover in *the* Pyrenean headwaters. In a changing climate, the trade-off between carbon sequestration and water uptake will be a key issue in the management of forest expansion (Varela et al., 2020). The accumulation of biomass also alters fuel load and behavior, which becomes more relevant on areas with Mediterranean-like climate. In some valleys of *Ordesa*, for instance, some small wildfires have arrived close to the park's boundaries in recent years (Benito, 2012). With the expected changes in climate, wildfire risk will increase considerably, affecting areas where current risk is low or negligible.

5. Conclusions

The three National Parks analyzed exhibited similar and substantial processes of forest expansion in the last 60 years, which are explained by three processes, all of them related to the reduction of livestock during summer in subalpine areas: the colonization of abandoned crops and grasslands, the densification of open and sparse forest areas, and the aggregation of small forest patches into larger and more continuous forest areas. Despite some differences in the magnitude of changes between the three National Parks, we observed a common pattern of morphological and functional homogenization of the landscape, which points to an effect of land-use legacies. Mountain protected areas can be considered as complex social-ecological systems, and changes in their landscape configuration can affect not only biodiversity, but also the important ecosystem services they provide to surrounding areas and adjacent communities. These impacts can be positive or negative depending on the local context and the organisms affected, and there is an ongoing scientific debate about whether these effects should be minimized through active management, or we should let the revegetation process continue to contribute to the naturalisation of the landscape (Lasanta-Martínez et al., 2015). Moreover, different stakeholders may have diverse preferences, sometimes valuing desirable features in a landscape in different or even opposite ways (Frei et al., 2020; Gómez-Limón & Fernández, 1999). This debate is particularly relevant for protected areas, and more specifically for National Parks, where many of the services – including those that motivated their declaration – depend on the spatial variation of different land cover patterns (De Pablo et al., 2020). Whatever the stance adopted, the sustainable management of abandoned landscapes in protected areas requires the quantitative assessment of the extent of land-cover changes, the space–time patterns of abandonment, and its drivers.

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.landurbplan.2021.104240.

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