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How do the surrounding areas of national parks work in the context of landscape fragmentation? A case study of 159 protected areas selected in 11 EU countries

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ABSTRACT

Europe has unique natural values but also has the highest level of ecosystem fragmentation. Evaluating the effectiveness of protected area management is becoming an increasingly common practice. Our main goal was to assess the dynamics of LULC changes in the buffer zones of 159 national parks in 11 selected European countries on the basis of the CORINE Land Cover database in the period from 1990 to 2018. We used five landscape metrics in order to investigate whether high natural and landscape values in the areas surrounding national parks affect the degree and rate of landscape fragmentation and isolation. We checked the statistical significance of the differences in the measured values among different distances to parks (1, 2 and 3 km) in different years and countries using the two-way ANOVA test. Furthermore, the classical principal component analysis method was applied to measure data grouped by country and then averaged by year and distance factor. We showed that high natural and landscape values in the surrounding areas of national parks affect the degree and rate of landscape fragmentation. The patch density index shows an evident increase, both in dynamic terms (1990–2018) and, mainly, in the spatial aspect. In 2018, there was an increasing rate of the patch density index of approximately 5–7% compared to that in 1990. It should be kept in mind that management efforts focused on the buffer zones of national parks may have limited success. Undoubtedly, the obtained results will contribute to the development of landscape ecology and spatial developments in the context of the effective management of national parks and their surroundings.

1. Introduction

The latest evaluation of the European Environment Agency (EEA, 2020) shows that Europe's nature is in serious, continuing decline. Unsustainable farming and forestry, urban sprawl and pollution are the top pressures to blame for the drastic decline in Europe's biodiversity, threatening the survival of thousands of animal species and habitats (Gomes et al., 2020). Furthermore, rapid environmental changes in entire EU countries are often associated with various aspects of habitat fragmentation (Geneletti, 2006; Fahrig, 2003; EEA, 2012; EC, 2020; Dener et al., 2021). The main result of this process is that in Europe, the majority of areas with high natural value have become highly fragmented and isolated from one another, initiating serious consequences for the conservation of the species that use those areas (Tscharntke et al., 2002; Fahrig, 2003; Ewers and Didham, 2006; Bruschi et al., 2015; Ward et al., 2020). Additionally, most protected habitats and species are not in good conservation status, and much more must be done to reverse the situation (e.g., Antrop, 2004; Bastian et al., 2006; Kubacka and Smaga, 2019; Kubacka, 2019). This process represents a major concern for the effectiveness of biodiversity conservation and a key driving force of the loss of species (Fischer and Lindenmayer, 2007; Perpiña Castillo et al., 2018; Ustaoglu and Collier, 2018).

There is an urgent need for mechanisms that review available information and make recommendations to practitioners (Sutherland et al., 2004). For example, Aichi Target 11 (one of the actions that has been taken to enhance the implementation of the Strategic Plan for Biodiversity 2011–2020) states that all protected areas (PAs) should be effectively managed, and many countries have instituted processes

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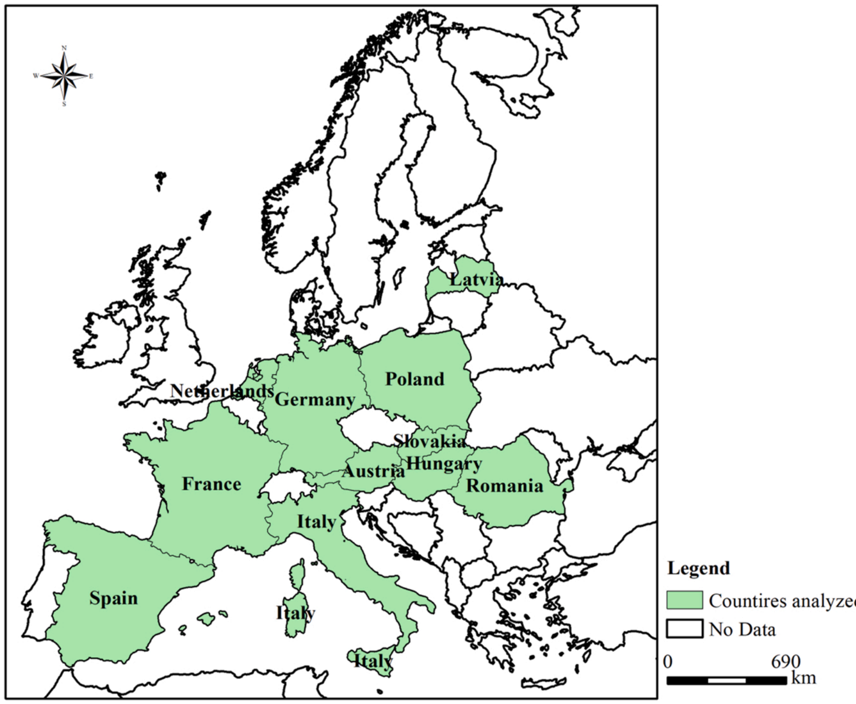

Fig. 1. Selected EU countries.

whereby management effectiveness is assessed; unfortunately, this target has been only partially achieved (IPBES, 2019; EC, 2020; SCBD, 2020).

Today, the most important issue in the protection system is providing policy-makers at the local, regional and national scales with the best possible information about the value of PAs; the communication of this information is very important for preserving biodiversity and ecosystem services properly and, also very important, supporting human quality of life. All PAs and all world ecosystems need to be effectively managed, and all protected areas and all nature need to be well-connected systems; this constitutes our reason for studying not only the land use/land cover (LULC) changes inside PAs but also their surroundings.

For a national park (IUCN Category II), its size and its main objective are aimed at protecting functioning ecosystems. A national park is defined as a "large natural or near-natural area protecting large-scale ecological processes with characteristic species and ecosystems, which also have environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities" (Shadie and Dudley, 2013, pp.16). The main objective of national parks is "to protect natural biodiversity along with its underlying ecological structure and supporting environmental processes, and to promote education and recreation" (Shadie and Dudley, 2013, pp. 16).

In the European Union, national parks (NPs) represent approximately 13% of the total protected natural areas, including approximately 25% of the areas that are protected by the EU, national PAs and Natura 2000 sites (EEA, 2012). The surrounding areas of an NP may be designated for consumptive or nonconsumptive use but should nevertheless act as a barrier for the defense of the protected area's native species and communities to enable them to sustain themselves in the long term (Borrini-Feyerabend et al., 2013).

The study of the ecological effectiveness of PA management is becoming an increasingly common practice, and these studies have focused mainly on either changes within boundaries or their ecological connections (e.g., Bruner, 2001; Gaston et al., 2008; Cook et al., 2014; Geldmann et al., 2015; Barnes and Turner, 2016; Gray et al., 2016), measured in terms of habitat loss and quantified by the perceived avoided losses of habitat (e.g., forests, grasslands, wetlands) within parks (e.g., Figueroa and Sánchez-Cordero, 2008; Nagendra, 2008) and in surrounding buffer zones (e.g., Nagendra, 2008; Leroux and Kerr, 2012). Most studies of PA effectiveness have focused on changes in forest cover (Geldmann et al., 2013) and analyzed changes and drivers on large spatial scales (Green et al., 2013). There is a need for studies evaluating the effectiveness of conservation actions with even stronger considerations of the limited budgets available for conservation and the ever-increasing rates of environmental change (Pullin and Knight, 2001; Sutherland et al., 2004).

Some studies have shown that, globally, national parks effectively reduce forest loss, although many are becoming increasingly isolated by high rates of forest loss in surrounding areas (Defries et al., 2005; Joppa et al., 2008; Joppa and Pfaff, 2011). Therefore, in many countries, buffer zones have been created that strictly surround the borders of NPs and are designed individually for each park to protect it against external threats resulting from human pressures. Understanding land-cover changes near protected areas is critical to ensure the resilience of the global network of protected areas (Martinuzzi et al., 2015).

Measuring PA effectiveness is not a simple task (Ferreira et al., 2020). Due to the number of metrics that could be used and, most importantly, the challenge of obtaining accurate data on these metrics, there is a limited understanding of the extent to which PAs deliver positive biodiversity outcomes (e.g., Coetzee et al., 2014; Coad et al., 2015). On the other hand, landscape metrics are helpful in research because they allow for the assessment of structural changes in patches of individual ecosystem types, as well as the diagnosis of the structural and functional relationships among patches within coverage classes and between classes throughout entire landscape units (McGarigal et al., 2012). Comparing landscape metrics from different periods can help quantify the changes that are taking place in landscapes, including the diversity of a landscape, the degree of its fragmentation, the spatial isolation of ecosystems, the disappearance or increase of their surfaces and other factors (e.g., Botequilha et al., 2006; Walz, 2011; Marshal et al., 2020).

The aim of this study was to assess the dynamics of LULC changes in the nearest surroundings of 159 national parks in 11 selected EU countries. We hypothesized that the high natural and landscape values

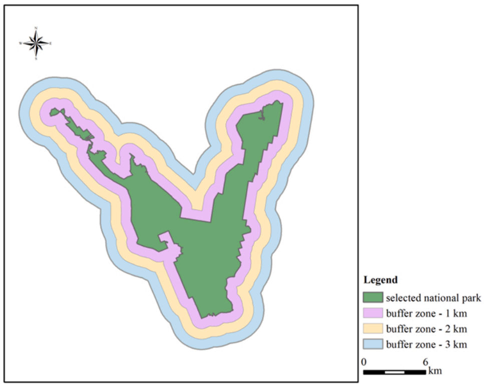

Fig. 2. The visualization of selected buffer zones.

of national parks may cause negative LULC changes in their nearest surroundings, which may further result in changes in the degree and rate of landscape fragmentation and the isolation of the parks themselves.

2. Materials and methods

This paper assesses the state of LULC changes in the surrounding areas of national parks using an analysis of the CORINE Land Cover (CLC) database on 159 national parks from 11 EU countries (Fig. 1).

The first criterion used to select countries was the number of national parks in the country that the International Union for Conservation of Nature (IUCN) classified into category II (national parks). We decided that the minimum number of national parks in each of the analyzed countries must be at least 4 to ensure that this category plays a significant role in the nature conservation policies in these countries. The second criterion was access to the CLC database in all study periods:

Table 1
Selected landscape metrics.

Metric name	Indicator abbreviation	Formula	Units
Patch Density	PD	$PD = \frac{n_i}{A}(\mathbf{10.000})(\mathbf{100})\mathbf{n_i}$ = number of patches in the landscape of patch type (class)$\mathbf{A}$ = total landscape area (m^2)	Number per 100 ha
Landscape Shape Index	LSI	$LSI = \frac{e_i}{\mathbf{min}e_i}\mathbf{e_i}$ = total length of edge of class in terms of number of cell surfaces$\mathbf{min\ e_i}$ = minimum total length of edge of class in terms of number of cell surfaces	NoneLSI ≥1, without limit.
Largest Patch Index	LPI	$\mathbf{LPI} = \frac{\mathbf{maxn}_{j=1}(\mathbf{a_{ij}})}{\mathbf{A}}(\mathbf{100})\mathbf{a_{ij}}$ = area (m^2) of patch ij$\mathbf{A}$ = total landscape area (m^2)	Percent0 < LPI ≤100
Landscape Division Index	DIVISION	$\mathbf{DIVISION} = \left[1 - \sum_{j=1}^{n}\left(\frac{a_{ij}{}^2}{A}\right)\right]\mathbf{a_{ij}}$ = area (m^2) of patch ij$\mathbf{A}$ = total landscape area (m^2)	Proportion0 ≤ DIVISION < 1
Modified Simpson`s Diversity Index	MSDI	$\mathbf{MSIDI} = -ln\sum_{i=1}^{m}P_i^2\mathbf{P_i}$ = proportion of the landscape occupied by patch type (class) i	NoneMSIDI ≥ 0, without limit

Source: McGarigal and Marks (1995).

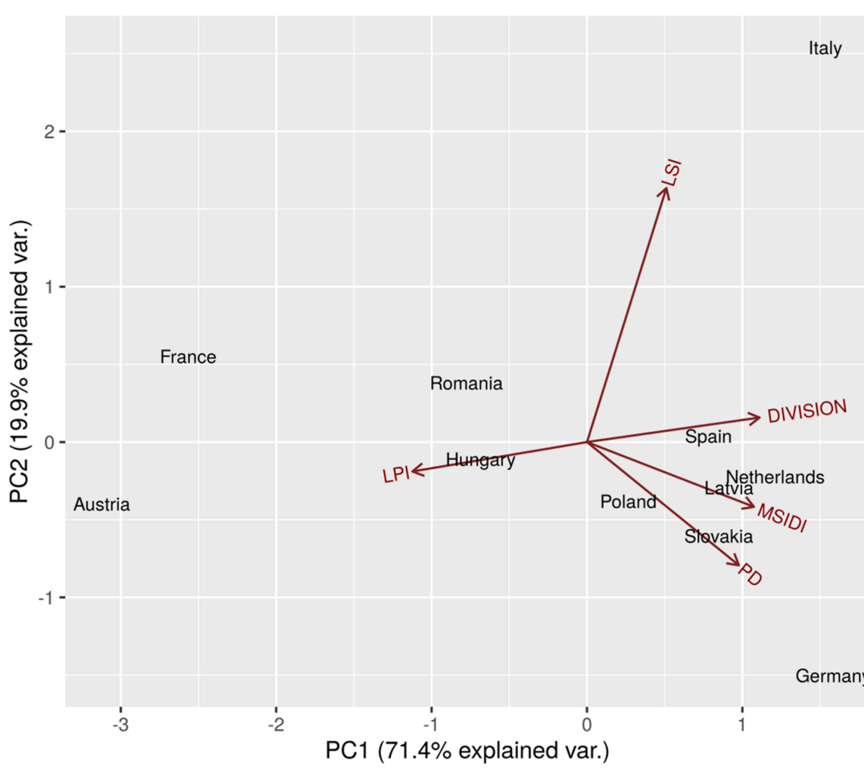

Fig. 3. A graph of the variability distributions of the analyzed landscape metrics (PD, LPI, LSI, DIVISION and MSIDI) in each of the selected countries.

1990, 2000, 2006, 2012, 2018. Only EU countries for which information in the CLC database was available from 1990 to 2018 were selected as test areas. Finally, when applying these nomination conditions, the total number of selected countries was 11 (see Fig. 1).

We delineated three concentric 1 km-wide polygons around each national park (NP) with ArcGIS version 10.7.1 (1, 2 and 3 km from the NP border; see Fig. 2) and used these sampling units to test the effectiveness of park effects on the surrounding LULC to a maximum distance of 3 km from the park boundaries. The selection of zones was justified by the analysis of the existing buffer zones in some national parks. The widths of buffer zones are usually not theoretically justified and depend on the individual characteristics of each national park. In our case, we estimated the width of the buffer zones based on the analysis of designed buffer zones in several tested national parks. Additionally, we wanted to assess the LULC changes and show the dynamics of landscape fragmentation and isolation in the immediate surroundings of each park to illuminate the impact of the park on the pace of these changes.

To evaluate the LULC changes in the areas surrounding the national parks, a set of metrics at the landscape-level scale were chosen (Table 1). We selected landscape metrics that can be easily interpreted by different groups of recipients and, at the same time, provide an overall assessment of the states of the analyzed areas. Initially, five indicators were selected: patch density (PD), largest patch index (LPI), landscape shape index (LSI), landscape division index (DIVISION) and modified Simpson's diversity index (MSIDI). The metrics used for the analysis are of a universal nature and they are commonly used in landscape research (e.g. Alves et al., 2021; Getu and Bhat, 2021; Kubacka and Smaga, 2019; Kubacka, 2019; Lamine et al., 2017). The selection of landscape metrics was also dictated by the choice of database.

After conducting a preliminary statistical analysis of the results obtained from FRAGSTATS, it was decided that only three of the metrics would be further analyzed. The PD, DIVISION and MSIDI indicators measure the same type of information which can be observed thanks to principal component analysis. Projecting them into two first principal components (which explain more than 90% of total variability), shows that they are highly correlated (Fig. 3). As such they will not give more insight into the structure of data analysed further with the use of regression models. Therefore, only three indicators were further analyzed (PD, LPI and LSI) because they provide consistent data with which to evaluate the fragmentation process.

The patch density (PD) metric is used to evaluate the degree and dynamics of landscape fragmentation, which is strongly reflected in the state of habitats (e.g., Bastian et al., 2006; Lai et al., 2017). Shape metrics, rather than patch area or isolation metrics, generally have a weak correlation with habitat amounts and may therefore be suitable for distinguishing between the effects of habitat amounts and configurations (Wang et al., 2014). This group of metrics contains a number of different indices that can be used to describe the geometric complexity of patch shapes at the patch, class or landscape level. Within this group, the landscape shape index (LSI) quantifies the number of edges of a given land cover class relative to that of a maximally compact and simple shape (i.e., a circle) of the same area, capturing several configurational changes associated with the division of the habitat (Saura and Carballal, 2004). Therefore, the LSI metric is likely to be biologically meaningful for species that are sensitive to habitat edges. The landscape shape index (LSI) provides a standardized measure of the total edges or the total edge density that adjusts to the size of the landscape. Furthermore, it has a direct interpretation, and if the LSI metric increases, it means that patches have become increasingly disaggregated. The third selected indicator was the largest patch index (LPI), which was considered an important indicator at the stage of assessing the degree of landscape and environmental fragmentation (McGarigal et al., 2012). Measuring diversity has been explored across landscape ecology with various metrics. Landscape division index (DIVISION) is based on the cumulative patch area distribution and is interpreted as the probability that two randomly chosen pixels in the landscape are not situated in the same patch (McGarigal and Marks, 1995). The last tested metrics is Modified Simpson`s Diversity Index (MSIDI). In ecology, it is often used to quantify the biodiversity of a habitat. MSIDI eliminates the intuitive interpretation of Simpson's diversity index (SIDI). This diversity index was initially developed as a way to calculate the richness of a particular species in an area, and is still widely used for that purpose (Comer and Greene, 2015).

Table 2
Coefficients used in linear regression analysis.

Coefficient	The equation used for the regression	Explanations
1) the slope of the regression curve of the measure with respect to the year and per distance, measuring the variability over time (3 coefficients)	$\mathbf{M}_{d_i}(\mathbf{year}) = \boldsymbol{\alpha}_{\mathbf{M}}^{\mathbf{d}_i} \times \mathbf{year} + \boldsymbol{\beta}_{\mathbf{M}}^{\mathbf{d}_i}$	**M** – measure (M = PD, LSI, LPI, DIVISION, MSIDI)$\mathbf{d_i}$ – distance (d_i =1, 2, 3 km)$\boldsymbol{\alpha}_{\mathbf{M}}^{\mathbf{d_i}}$ - regression slope for measure **M** and distance $\mathbf{d_i}\boldsymbol{\beta}_{\mathbf{M}}^{\mathbf{d_i}}$ - regression intercept for measure **M** and distance $\mathbf{d_i}$**M** – measure (M = PD, LSI, LPI, DIVISION, MSIDI)$\mathbf{p_i}$ **–** period (p_i = 1990, 2000, 2006, 2012, 2018)E_M– regression slope for measure $\mathbf{M}\gamma_M$– intercept**C** - country
2) the averaged slope of the regression curve of the measure relative to the year, measuring the variability over time (1 coefficient)	$\boldsymbol{\beta}_{\mathbf{M}} = \frac{\sum_{i=1}^{3} \boldsymbol{\alpha}_{\mathbf{M}}^{\mathbf{d}_i}}{3}$	
3) the average value of the measure in each country per distance, measuring the general differences in the values of indicators among countries (3 coefficients)	$\mathbf{C}_{d_i} = \frac{\sum_{i=1}^{5} \mathbf{M}_{d_i,\ \mathbf{p}_i}}{5}$	
4) the average value of the measure in each country, measuring the general differences in the values of indicators among countries (1 coefficient)	$\mathbf{D} = \frac{\sum_{i=1}^{3} \mathbf{C}_{\mathbf{d}_i}}{3}$	
5) the average slope of the measure's regression curve in relation to the distance, measuring whether the measure increases or decreases with the distance in each country (1 coefficient)	$\mathbf{M}(d_i) = \mathbf{E}_{\mathbf{M}} \times \mathbf{d}_i + \boldsymbol{\gamma}_{\mathbf{M}}$	

Source: Own works.

The current standards of the national parks i.e. as of 01.06.2020 was used in the geostatistical analysis. The data source for the selected national parks was the World Database on Protected Areas (https://www.protectedplanet.net/c/world-database-on-protected-areas), which is the most up-to-date and complete source of information on protected areas and is updated monthly with submissions from governments, nongovernmental organizations, landowners and communities. Obviously one should bear in mind that these standards are constantly updated. Moreover, some national parks have been established in the period in which the analysis was carried out (1990–2018) as well as the IUCN categories are constantly updated.

It is managed by the United Nations Environment World Conservation Monitoring Centre (UNEP-WCMC, 2018) with support from IUCN and its World Commission on Protected Areas (WCPA). The source of the LULC map was the CORINE Land Cover database, which consists of an inventory of land cover in 44 classes (https://land.copernicus.eu/pan-european/corine-land-cover). The CLC database uses a Minimum mapping unit (MMU) of 25 ha for areal phenomena and a minimum width of 100 m for linear phenomena. Its spatial and temporal limitations can affect the obtained results (e.g., Rodriguez-Rodriguez and Martinez-Vega, 2017; Kubacka, 2019; Kubacka and Smaga, 2019; Rodriguez-Rodriguez et al., 2019a, 2019b).

We used the FRAGSTAT 4.2.1 program to calculate a set of landscape metrics for characterizing changes in LULC across five time periods: 1990, 2000, 2006, 2012, and 2018. Therefore, this study represents

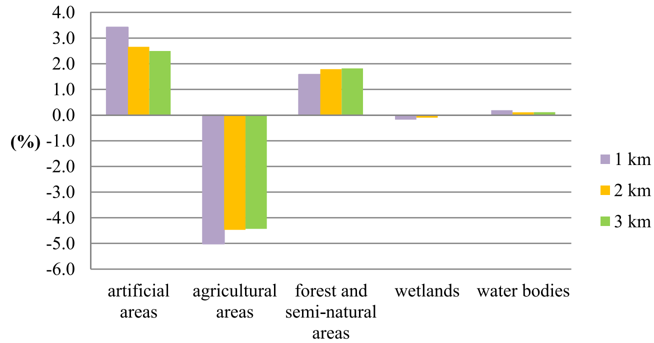

Fig. 4. Average percentage of landscape (PLAND) changes (%) in all selected national parks by CLC classes in three buffer zones for the overall study period (1990–2018).

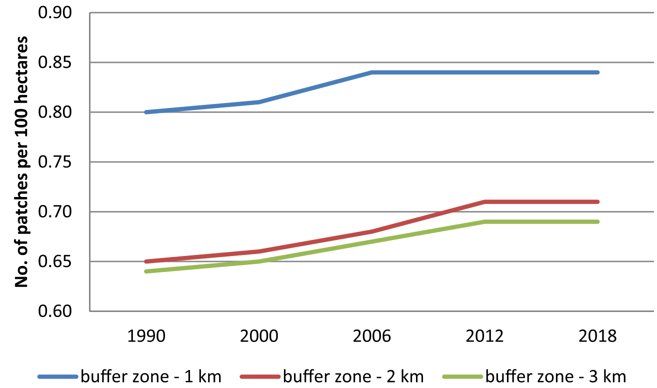

Fig. 5. The average patch density (PD) in all analyzed national parks.

close to 30 years of landscape change analysis in the longest time period available: 1990–2018. All GIS analyses were performed using ArcGIS 10.7.1 software.

In the first step, we visualized the subsets of the data split by each variable (facet plot). In each subplot, we plotted the linear regression curves for the given subsets. Then, we checked the statistical significance of the differences in the measured value between distances to parks in different years and countries using the two-way ANOVA test (Chambers et al., 1992). When the p value was smaller than or equal to the significance level α (0.05), we rejected the null hypothesis and noted significant differences. The classical principal component analysis (PCA; Venables and Ripley, 2002; R Core Team, 2019) method was applied to measure data grouped by country and then averaged by year and distance factors. This method allowed us to observe the natural clusters in the spatial components of the analyzed measures.

Based on the visualization, an assumption can be made about the linear nature of changes in the values of individual measures with respect to the year and distance. One-way linear regressions were carried out, allowing us to obtain the following coefficients for each country and measure (Table 2).

Principal components were analyzed again with these data, this time considering the slope coefficients, i.e., the variability depending on the distance and year. In this way, it was possible to construct a more complete visualization space.

The grouping of countries based on the raw figures is not correct due to the temporal variability in the data. In such a case, the analysis of the raw data would be burdened with too much uncertainty. An analysis of the averaged slope coefficients of 3 selected measures was applied (according to the conducted PCA).

All statistical analyses were performed using R software (R version 3.6.3; R Core Team, 2019).

3. Results and discussion

At the global scale, it has been estimated that human populations increase on the borders of protected areas (e.g., Wittemyer et al., 2008; Joppa et al., 2009), and our analyses have reached the same conclusions. In the period from 1990 to 2018, the largest increase in anthropogenic areas took place in the 1st buffer zone (1 km), located closest to the national park borders (+3.42%). Along with moving away from the park borders, the share of this CORINE Land Cover dataset class decreased (Fig. 4). New artificial areas are being built on the sites of former agricultural areas. Some studies show that national parks effectively reduce forest loss (Barbier et al., 2010), although many are becoming increasingly isolated by high rates of forest loss in surrounding areas (Defries et al., 2005; Joppa et al., 2008; Joppa and Pfaff, 2011). Our analysis does not confirm this trend. On the other hand, in this CLC class, we can observe the effect of the borders of the national parks (Fig. 4). In the case of wetlands as well as that of water bodies, no significant changes were observed in any buffer zone or for the overall period (1990–2018).

The next step was to analyze the landscape structure. For this reason, three landscape metrics were calculated.

Patch density (PD) shows an evident increase (Fig. 5), both in dynamic terms (1990–2018) and mainly in spatial aspects (1, 2 and 3 km from the park borders). In 2018, there was an increase in the rate of the PD index by approximately 5–7% compared to that in 1990 in the first buffer zone (1 km). There was a linear upward trend that may cause further fragmentation of valuable natural and landscape areas in the near future. Much greater dynamics were noticed in the spatial aspect of the PD index. When moving away from the park borders, the fragmentation rate of the natural environment decreased. Comparing the values of the PD index among different buffer zones, those in the 2 km and 3 km buffer zones were 20% lower than those in the 1 km buffer zone.

This can pose some threat to the environment within the park itself, as the connections of matter circulation and energy flow are broken. In many altered landscapes, fragmentation is associated with a whole array of changes in biodiversity, community composition, abiotic conditions and biotic interactions (Fahrig, 2017). In addition, fragmentation feedbacks on connectivity may differentially affect specific groups, originating from the evolution of dispersal rates, syndromes and genetic diversity (Fahrig, 2003; Bonte et al., 2018). Our analysis confirmed that the closest areas to national parks are highly attractive for changing landscape dynamics, thus increasing fragmentation processes. All the time, we need to remember that the effectiveness of natural protection is strongly related to the state of the areas surrounding the protected regions. Unfortunately, high environmental values attract built-up areas and connect these built-up areas with technical infrastructure. In addition, in many cases, in the immediate zone bordering a national park, buffer zones are created whose task is to maintain ecological processes and the stability of ecosystems to preserve biodiversity in the area protected by the national park. As we can see, these buffer zones do not always fulfill their role.

Landscape or habitat fragmentation is the result of a gradual reduction in the natural-environmental surface as well as its progressive isolation (MacLean and Congalton, 2015). This process is one of the main threats to biodiversity (Haila, 2002; Jaeger et al., 2011; Bruschi et al., 2015; Fahrig, 2017). Generally, larger and well-connected ecosystems can better host and conserve local biodiversity than smaller and isolated ecosystems (Southerland, 1995). Furthermore, connectivity is a vital element of landscape structure and plays an important role in ecological dynamics within and among habitats (Bennett, 1990). The loss of surfaces has significant ecological implications for a wide array of taxonomic groups, including birds, mammals, reptiles, amphibians, invertebrates and plants (Fahrig, 2003; Fischer and Lindenmayer, 2007; Fletcher et al., 2007; Hilty et al., 2020), but not all species have the same sensitivity to habitat and landscape fragmentation (Aurambout et al., 2005; Cuervo and Møller, 2020). Landscape fragmentation is a process derived from both natural and anthropogenic forces. The main factor

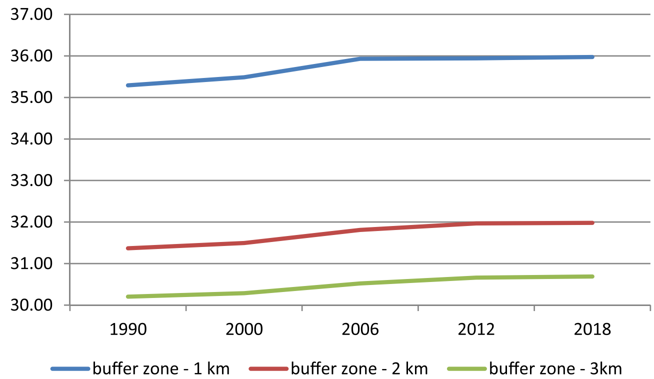

Fig. 6. The average landscape shape index (LSI) in all analyzed national parks.

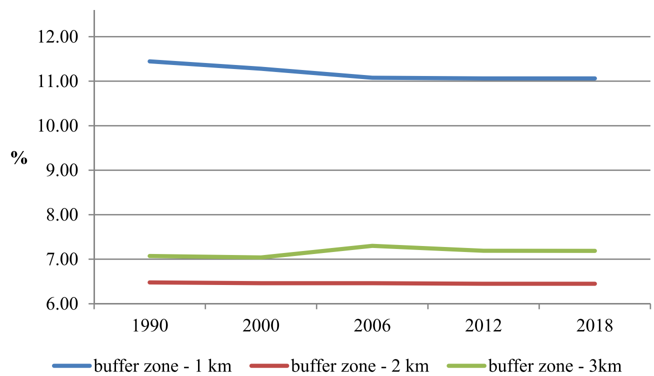

Fig. 7. The average largest patch index (LPI) in all analyzed national parks.

influencing landscape fragmentation is transportation infrastructure, which has been confirmed in numerous publications (e.g., Forman and Alexander, 1998; Geneletti, 2006; Pătroescu et al., 2007; Fahrig and Rytwinski, 2009; Macias and Gadziński, 2013). In fact, technical infrastructure development has led to the loss, modification and fragmentation of landscapes and natural habitats.

In terms of time, the LSI values slightly increased (Fig. 6), while there are clear differences in the spatial aspect among all buffer zones. The LSI values increased with the approach to the borders of all analyzed national parks (N = 159). The highest values were found in the 1st buffer zone (1 km), with an average of 35.72 in all analyzed periods. Then, the LSI value dropped to 31.72 in the 2nd buffer zone (2 km) and reached a value of 30.47 in the 3rd buffer zone (3 km). The largest difference was observed between the 1st and 2nd buffer zones, which amounted, on average, to 12.6% in all analyzed periods. Thus, the next indication clearly shows that the highest rate of the fragmentation as well as the disaggregation of patches with the same land use and land cover (LULC) class increases with the approach to the boundaries of the protected areas.

The largest patch index (LPI) quantifies the percentage of the total landscape area comprised by the largest patch (Fig. 7). The highest values were achieved in the first buffer zone (1 km; on average, 11.2%). While the degree of landscape fragmentation in the first buffer zone was much higher than those in the other analyzed zones, the first buffer zone still included some compact areas, mainly in the form of forest complexes. In terms of time, this indicator had a decreasing tendency, i.e., it can be concluded that the disaggregation phenomenon will intensify in the future, and even areas that have been aggregated so far may be fragmented. The other buffer zones had significantly lower LSI values (6.5% in the second buffer zone and 7.2% in the third buffer zone). This is a significant difference compared to the average value in the 1 km buffer zone (approximately 4.7%).

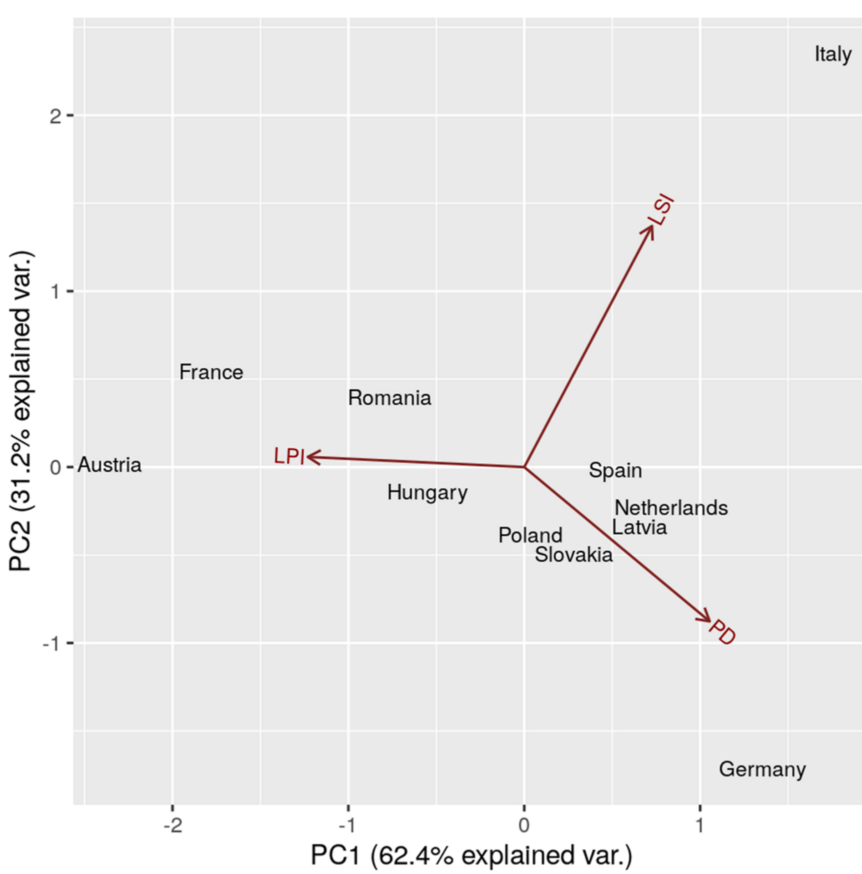

Fig. 8. A graph of the variability distributions of the analyzed landscape metrics (PD, LPI and LSI) in each of the selected countries.

Table 3
The statistical significance of the differences in the measure values among the distances to parks in different years and countries, determined using the two-way ANOVA test.

metrics and buffer zones	PD 1 km	PD 2 km	PD 3 km	LPI 1 km	LPI 2 km	LPI 3 km	LSI 1 km	LSI 2 km	LSI 3 km
year p-value	0.0648	0.0081	0.0182	0.4622	0.9711	0.2951	0.0111	0.0152	0.0383
country p-value	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

Source: Own works

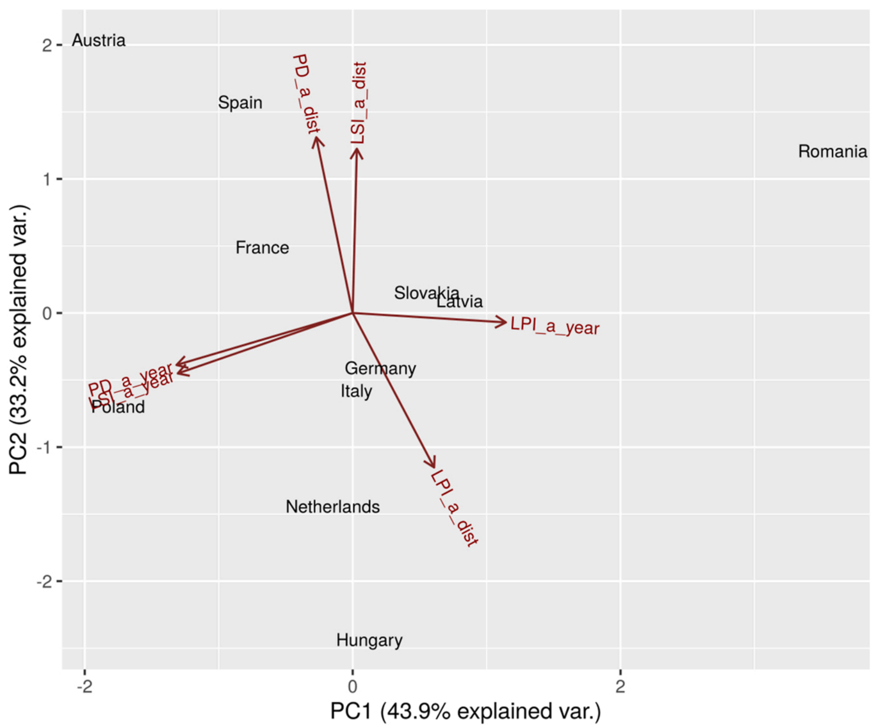

Fig. 9. A graph of the variability distributions of the analyzed landscape metrics (PD, LPI and LSI) in each selected country depending on the distance and year.

There are very strong linear trends in the analysis of all countries, both in terms of time (1990, 2000, 2006, 2012, 2018) and space (1, 2, 3 km). These trends can be divided in terms of the increase or decrease in the value of a given measure, but there is also a division into groups of analyzed countries (see Appendix 1 and 2).

A visualization was prepared to examine the dependence of the measure of and the distance from each national park more precisely (see Appendix 1). Moreover, each plot contained linear regression curves. For the spatial distribution, which was determined by the buffer zones surrounding all national parks (1, 2 and 3 km), it is clearly visible that most of the analyzed countries were characterized by the arrangement of values according to an increasing distance. Austria and Spain were an exception here. This is particularly evident for the PD indicator (see Appendix 2).

Statistical analysis also showed that the values of the PD metric explained 62.4% of all data variability, LPI - 31,2% and LSI only 6.4%

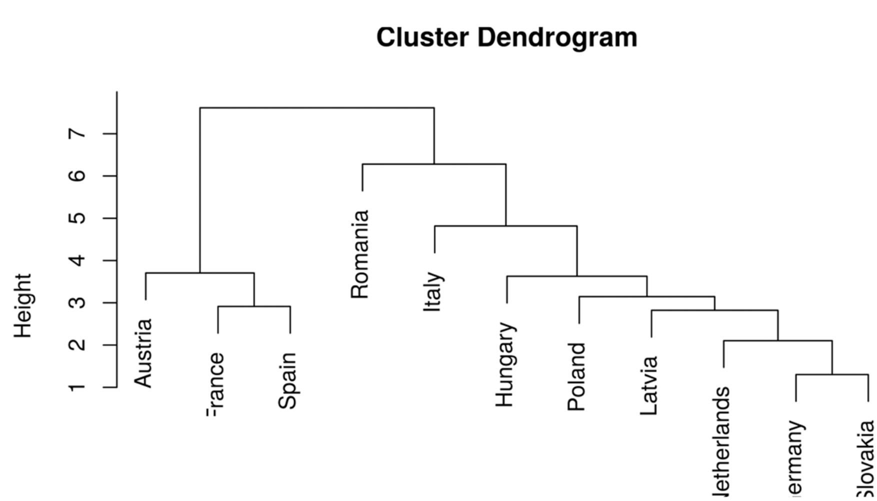

Fig. 10. Cluster dendrogram.

(Fig. 8).

The calculated landscape metric values demonstrate significant differences (p-value < 0.05) among all the analyzed countries in each year and within the countries among all buffer zones (Table 3).

Then, one-way linear regression was carried out, which allowed us to obtain the slope coefficients of the regression curve for each country and each of the analyzed measures, i.e., the variability depending on the distance and year. In this way, a perfect division of space and a confirmation of the significance of the selected metrics were obtained. Each of these components had a different character, and the distinctions among them were very significant (Fig. 9 and Table 3). We can confirm that the PD and LSI indicators in the regression with respect to the year are identical, while the PD and LSI indicators are identical in terms of distance. Another interesting topic for the further research would be the problem of integrating multiple indicators like (PD, LPI and LSI) into one simple, unified measure that will sustain the properties and interactions among original input indicators. This may be achieved using known aggregation methods or new interaction-driven (Żywica et al., 2021).

The last statistical analysis included an attempt to group countries according to the values of the individual indicators. Initially, an attempt was made to group the original data for all analyzed countries (N = 11). Due to the high temporal variability in the data, this analysis was subject to high uncertainty. Therefore, the averaged data were used. The dendrogram clearly shows that among the analyzed countries, two groups can be distinguished. The first group included Austria, France and Spain, and the second included the other countries. The first group was characterized primarily by low PD indicator values, which increased with time and distance. Therefore, in their case, the degree of fragmentation increased with distance from the borders of the national parks (Fig. 10). There may be several reasons for these grouping results. The first and most important driving force arising there from is that each analyzed country had a completely different history of environmental and landscape protections, which may constitute the main aspect of the effectiveness of the protective actions taken up. The second important factor is economic development, which can clearly affect the dynamics of LULC changes. Other factors that should be taken into account in further analysis include the location of the analysis object against the background of the biogeographical division of Europe, the distance to the nearest highly urbanized area and technical infrastructure development.

4. Conclusion

The system of environmentally valuable areas on local, regional, national and international scales should be considered in terms of ecological (natural) networks. No area, even the smallest area with regard to the environmental value, regardless of whether it is under protection or not, can function properly without connection with other environmentally valuable areas. This connection must be functional enough to not disturb the migration of matter and energy. That is why it is so important to monitor changes in land cover and use not only within protected areas themselves but also in their close surroundings. Our results suggest that the selected national parks (N = 159) in 11 EU countries were largely effective in avoiding LULC changes within their closest surroundings in all analyzed periods (1990–2018). On the other hand, the largest changes in the structure of the landscapes occurred in the first buffer zone (within 1 km) from the park boundaries, increasing the negative impacts of one of the most problematic landscape dynamics in terms of nature conservation: fragmentation and loss of structural connectivity. This issue is especially critical in the context of climate change, and for these reasons, the maintenance and recovery of connectivity became mandated by international conservation targets such as Aichi Target 11 of the Convention on Biological Diversity.

National parks are a key figure of nature and landscape protection in Europe and throughout the world, so it is necessary to improve the guaranties of conservation thereof. In this context, the reduction in landscape fragmentation and the improvement of the connectivity among protected natural areas are crucial, especially under the impact of climate change. With this objective, we need to pay special attention to all the LULC changes in the areas surrounding national parks to avoid the fragmentation process and to achieve the objectives of the EU Biodiversity Strategy for 2030 (Hilty et al., 2020).

Therefore, it is necessary to design and implement relevant policy decisions to strictly regulate buffer zones around national parks and reduce the influence and impacts of the activities and LULC dynamics in the surrounding areas. The objectives are to improve the spatial management and conservation of natural heritage and ensure the maintenance of connectivity, avoiding fragmentation processes not only inside national parks but also in their buffer zones. The results obtained confirm that the problem of increasing fragmentation of the environment and, consequently, decreasing degrees of integrity of the entire ecological system affects most of our case studies; therefore, we should strive to develop European guidelines for the planning and spatial development of the national park buffer zone and then to implement relevant provisions that allow for their proper implementation into protection laws by each member state. On the other hand, preference should be given to appropriate types of land use, including minimizing environmental barriers preventing or impeding the flows of matter, energy and biological information.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.landusepol.2021.105910.

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