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Assessment of Endemic Lycian Salamanders Habitats Impacted by 2021 Mega Forest Fires in Turkey

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Abstract. The Lycian salamanders consist of seven allopatric endangered and endemic species restricted to Mediterranean Turkey and some adjacent Aegean islands. Mega forest fires occurred consecutively over a prolonged period of time in the distribution areas of six of the species in 2021, leading to habitat fragmentation and habitat loss. In this study, we find that a total of 751.9 hectares of Lycian salamander habitats were lost due to the mega-fires that occurred in summer 2021. Our results suggest that *L. atifi* is the most vulnerable with a loss of 285.84 hectares of habitat area, followed by *L. flavimembris* with 242.54 hectares, and *L. antalyana* with 124.16 hectares. *L. fazilae* is the species which suffered the least habitat loss, at 25.83 hectares. *L. billae* and *L. luschani* suffered habitat losses of 30.83 and 43.40 hectares respectively. When the transformation of morphological classes was examined, a significant decrease of all species was observed in the core areas which ensure spatial connectedness, and the edge magnitude, which was taken as an indicator of fragmentation, increased. Bridges providing connectedness were observed to have increased for some species. This indicates that while existing connections in habitats were fragmented due to the forest fires, potential connections may be formed after the forest fires. When the fragmentation values were examined according to the results of pattern analysis, the most notable marginal increase in fragmentation before and after the fires was found to have occurred in the habitat of *L. atifi*. In addition, we discuss recommendations for the sustainability of species populations in habitat restoration and forest management.

Key words: *Lyciasalamandra*, wildfire, habitat loss, spatial pattern, southwestern Anatolia.

Introduction

Amphibians are among the most vulnerable organisms on Earth and changes in land use and global warming constitute the most critical threats to their existence (Hof et al., 2011; Nelson et al., 2021). Forest fires, one of the causes of habitat loss, may affect caring capacity and population density by damaging habitats. After habitats have been fragmented, the inability of species to change their home ranges may cause the local density to drop below caring capacity and reduce their survival rates (Lappan et al., 2021). The disturbance of ecological processes due to forest fires also poses a threat to biological diversity (Driscoll et al., 2021). Only a few studies have been undertaken on the effects of forest fires on salamander populations, particularly following forest fires (Renken, 2006; Greenberg & Waldrop, 2008; Ford et al., 2010; O'Donnell & Semlitsch, 2015; Margolis & Malevich, 2016). The direct and indirect impacts of forest fires on terrestrial salamanders are assumed to be affected by a variety of factors including seasonality, the frequency of the forest fires, the intensity of the forest fires and the historical fire regime (Greenberg & Waldrop, 2008). While (Stebbins & Cohen, 1995) argue that amphibians' humid and permeable skin and eggs make them more vulnerable to heat and the drying up of microhabitats, (Komarek, 1969) claims that amphibians do not seem to be uncomfortable near fire and respond with adaptive behaviour that reduces their death rate.

Forest fires are historically the primary form of disruption in the scrub and coniferous forests along the Mediterranean coast. Due to the region's various physiographical and climatic characteristics and the accumulation of combustible materials, fire regimes also vary. The Mediterranean has witnessed its highest temperatures and lowest humidity in the last ten years likely as a result of climate change, and has experienced drought in the last two years. There was a tendency in the region towards an increase in the frequency and severity of forest fires, and – again within the last two years – there have been predictions of mega forest fires (Jiménez-Ruano et al., 2017; Moreira et al., 2020). There are grave concerns that climatic factors such as extreme temperatures and prolonged periods of drought along the European Mediterranean coast will extend the forest fire season in the near future (Cardil et al., 2014). In order to understand the effects of forest fires on endemic species and to monitor and assess ecological processes, spatial relationships need to be measured in relation to short- or long-term changes over time (de Vries et al., 2003; Corona, 2016). Remote sensing, geographic information systems (GIS) and the integration of landscape metrics combine to provide the detailed, spatially consistent information required for ecosystem services, sustainable resource management and land use planning (Kayiranga et al., 2016; Dutta et al., 2020). Monitoring the fragmentation of forest habitats due to fires in the context of land use planning and sustainable environment management is important for preventing irremediable negative consequences (Hansen et al., 2013).

The Lycian salamanders (*Lyciasalamandra* (Veith & Steinfartz, 2004); Amphibia: Salamandridae) consist of the following seven allopatric species: *L. antalyana* (Başoğlu & Baran, 1976), *L. atifi* (Başoğlu, 1967), *L. billae* (Franzen & Klewen, 1987), *L. fazilae* (Başoğlu & Atatür, 1974), *L. flavimembris* (Mutz & Steinfartz, 1995), *L. helverseni* (Pieper, 1963), and *L. luschani* (Steindachner, 1891) (see (Sparreboom, 2014; Veith et al., 2016). These amphibian species represent an exceptional case of micro-endemism (Veith et al., 2020). They are all slender, terrestrial, and viviparous. The Lycian salamanders are to be found in limited areas and in relatively small patches. Endemic along the 385-kilometer stretch of the Mediterranean coast from the west of Mt. Menteşe to the southeastern Asar mountains and the southern Taurus mountains in Turkey, and on some of the nearby Greek islands in the Aegean (Veith et al., 2001; Öz et al., 2004; Eleftherakos et al., 2007; Franzen, 2008; Sparreboom, 2014; Göçmen & Karış, 2017), these salamanders live at altitudes of 40-1,150 metres on north-facing slopes (Franzen, 2008; Göçmen et al., 2013) and pine forest and maquis shrubland habitats (Veith et al., 2001; Rödder et al., 2011). The species have adapted to live deep within humid crevices in boulder fields at the foot of karstic limestone slopes (Sparreboom, 2014), since their habitats are hot and dry in summer, and warm and damp in winter. Within this naturally restricted range, the genus faces the risks of habitat loss as a result of forest fires and over-collection for scientific purposes. On the International Union for the Conservastion of Nature (IUCN) red list, two of the Lycian salamanders (*L. luschani* – (Steindachner, 1891)– and *L. helverseni* – (Pieper, 1963) are listed as vulnerable (VU). Four (*L. atifi* – Başoğlu, 1967, *L. antalyana* – Başoğlu & Baran, 1976, *L. fazilae* – Başoğlu & Atatür, 1974, and *L. flavimembris* – Mutz & Steinfartz, 1995) are listed as endangered (EN), and one (*L. billae* – Franzen & Klewen, 1987) is listed as critically endangered (CR). The populations of *L. flavimembris* in its northernmost limits and *L. antalyana* in its westernmost limits are on the decline (Kaska et al., 2009; Arslan et al., 2020).

Turkey's first mega fires occurred along the Mediterranean shoreline, the habitat of the Lycian salamander, in 2021. From now on, the Mediterranean basin will most likely experience mega fires more frequently due to global warming. This study, which focuses on the impact of the forest fires on the habitats accommodating the six species of Lycian salamanders with endemically limited distribution abilities, discusses (i) the extent to which salamanders were affected by the forest fires, (ii) the burn ratios of their habitats, and (iii) how ecological connectedness has changed before and after the forest fires. The results of this study reveal the effects of forest fires on the habitats of salamanders, which are endemically restricted at the local level. Finally, in addition to its findings, the study also discusses which areas attention should be paid to with respect to these species during the course of habitat restoration efforts in the wake of the forest fires.

Materials and Methods

Our study focuses on six Lycian salamanders (*L. antalyana, L. atifi, L. billae, L. flavimembris, L. fazilae, L. luschani*) that are restricted to the southwestern coast of Turkey (Fig. 1). The study area has a Mediterranean climate (Köppen: Csa) characterised by hot, dry summers and warm, wet winters. In addition, the temperature and precipitation may have effects on the local distribution ranges of the species. The study area is covered with scrubland and *Pinus brutia* forests, which primarily include *Olea europea, Juniperus oxycedrus, Phillyrea latifolia, Myrtus communis, Pistacia terebinthus, Quercus coccifera*, and *Arbutus andrachne*.

Field studies were carried out at intervals in November–February between 2010–2020 and all records of the species were confirmed from previously published literature (Başoğlu & Atatür, 1974; Mutz & Steinfartz, 1995; Veith et al., 2001; Öz et al., 2004; Johannesen et al., 2006; Beukema et al., 2009; Akman et al., 2011; Göçmen & Akman 2012; Göçmen et al., 2013; Akman & Godmann, 2014; Üzüm et al., 2015; Göçmen & Karış, 2017; Arslan et al., 2018; Oğuz et al., 2020; Veith et al., 2020). A total of 249 presence records were gathered for the six species. In all, 62 records were obtained for *L. antalyana*; 35 for *L. atifi*; 35 for *L. billae*; 24 for *L. fazilae*; 42 for *L. flavimembris* and 51 for *L. luschani*. Where locality information lacked coordinates, it was referenced to the closest location provided in earlier studies using Google Earth Pro v. 7.1.5 (Google Inc.). All records were georeferenced using a WGS84 coordinate system and their accuracy was checked with ArcGIS (v10, ESRI, California, USA). The extent of occurrence (EOO – (Joppa et al., 2016) of each species was calculated, to establish its distribution, with the help of the IUCN Red List Toolbox for ArcMap. The EOO measures the spatial spread of the areas currently occupied by the taxon. It is ascertained by applying a Minimum Convex Polygon (as in the IUCN

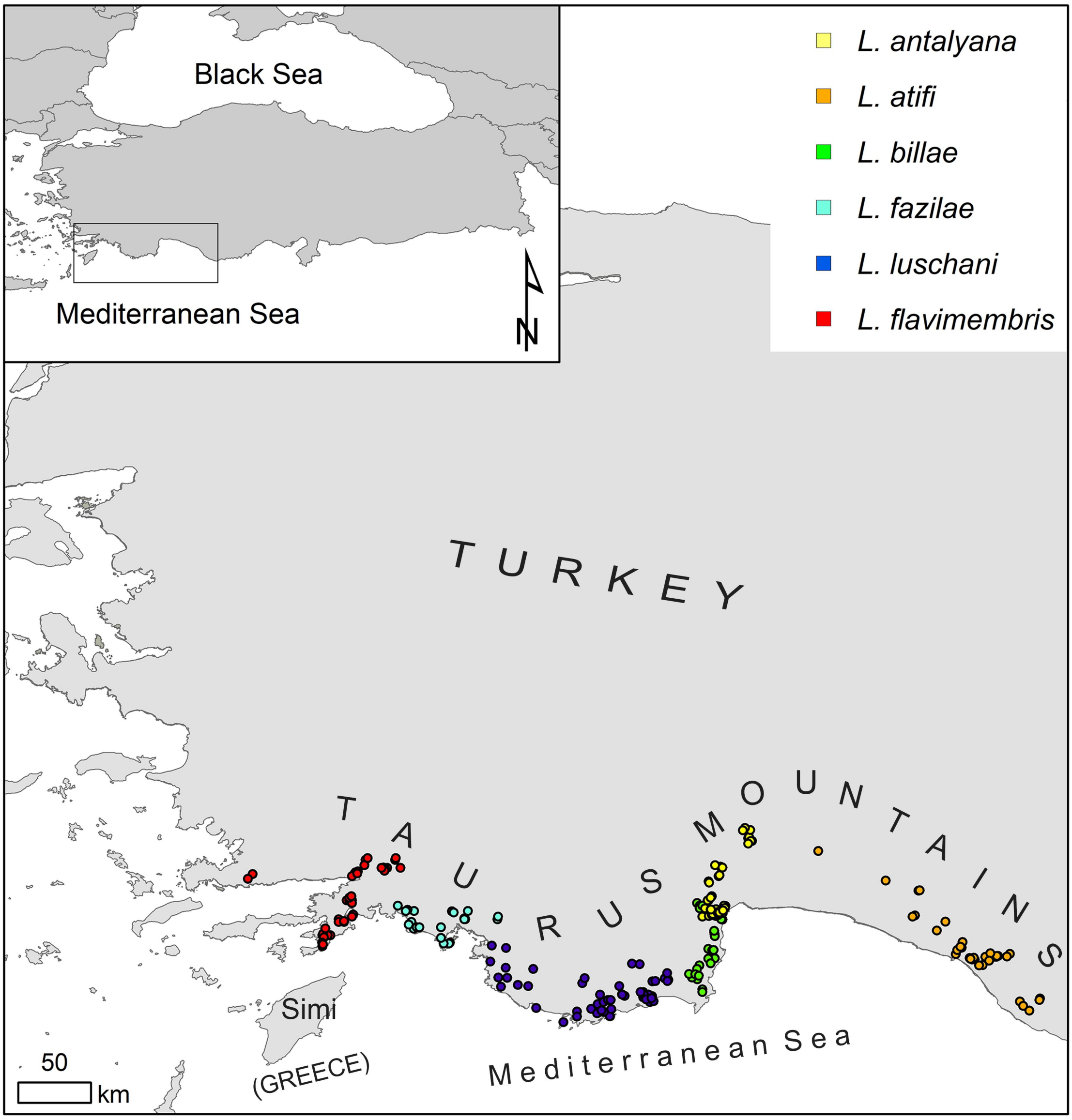

Fig. 1. The distribution of Lycian salamanders in south-western Anatolia.

Red List Criteria) to measure how far risks from threats are spread spatially across the geographical range of the taxon (Bland et al., 2017; IUCN Standards and Petitions Committee 2019).

The methodology for the study consists of three parts – namely, identifying habitat losses in accordance with burn rates based on the calculation of spectral indices, examining transformations in morphological classes using Sankey diagrams, and measuring the degree of fragmentation. Spectral indices (NDVI and NBR) were measured in the R 4.0.3 environment. The degree of fragmentation was determined using the GuidosToolbox (Graphical User Interface for the Description of image Objects and their Shapes) software. Here, we used Level-1 C products of Landsat-8 OLI data with 30 m spatial resolution that was downloaded from the U.S. Geological Survey (USGS) website. The dates when the vegetation was still active were taken into consideration while downloading satellite images before the fire, and a total of 12 pre-fire and post-fire images with a cloud cover less than 2 per cent were downloaded. Since cloudy areas did not overlap with forest areas, masking was not necessary. The Landsat 8 observation satellite system has two different sensors: the Operational Land Imager (OLI) and the Thermal Infrared Sensor (TIRS). With the help of these sensors, 11 different wavelength images were captured. The satellite system also has calibrated and scaled Digital Numbers (DN). These numbers are represented in 16-bit unsigned integers and using the radiometric re-scale factors from Landsat metadata files (MTL), can be re-scaled according to Top of Atmosphere (ToA) reflectance values.

During the initial stage, surface reflectance correction was carried out using ToA and dark object subtraction (DOS) for the radiometric calibration of satellite images. ToA corrections often result in negative reflectance values. Landsat atmospheric correction and surface recovery algorithms are not ideal for bodies of water due to the naturally low level of water brightness and the resulting low signal-to-noise ratio.

After radiometric calibration, the bands were merged. Bands 4, 5 and 7 were merged and the obtained raster map was framed according to boundaries of the study area. Then the Normalized Difference Vegetation Index (NDVI) was calculated. NDVI uses the red (R) and near infrared (NIR) values, since energy reflected in those wavelengths is correlated with the amount of surface vegetation. NDVI values are standardised and vary between -1 and +1. NDVI values which are greater than 0.3 are assumed to represent vegetation.

Pixels that do not represent vegetation (< 0.3) can be masked out (Bhandari et al., 2012; Gandhi et al., 2015). In R, the "*calc*" function is used for arithmetic calculations using raster data (in addition to other applications). In that case, a function is written to select all values smaller than 0.3 and to mark these values as n/a (not applicable). This way, the value of the image acquired will be bigger than 0.3. This operation is called "thresholding".

The Normalised Burn Ratio (NBR) is used to define the burned areas. This index uses near the infrared (NIR) band where the vegetation has high absorption levels and the short-wave infrared (SWIR) band for damaged woody vegetation with higher reflectance values. The NBR is calculated as follows:

$$\mathrm{NBR} = \frac{\mathrm{NIR}-\mathrm{SWIR}}{\mathrm{NIR}+\mathrm{SWIR}}.$$

The NBR index was originally developed for use with the Landsat 4-5 Thematic Mapper (TM) and the Enhanced Thematic Mapper Plus (Landsat 7 ETM +). However, it works with any multispectral sensor with an NIR of band between 760-900 nm and a SWIR band between 2080-2350 nm (Cocke et al., 2005). As a result, the SWIR2 band is used for Landsat 8.

The Normalised Burn Ratio (dNBR) is better suited to understanding the scope and severity of forest fires when used after calculating the differences in pre-fire and post-fire conditions. This difference is best measured by collecting data just before and after the fire (Miller & Thode, 2007):

$$\mathrm{dNBR} = \mathrm{prefire\ NBR} - \mathrm{postfire\ NBR}.$$

The burn severity map is categorised according to the table below (Table 1). It is important to underline that these values, which quantitatively display burn severity, may represent various outcomes and be interpreted differently.

After forming the dNBR map, the areal distribution of pixel values was calculated by burn severity levels, and the change in the number of habitat areas following the forest fires was examined.

In the second stage, presence/absence data was formed (vegetation 1, no vegetation 0) using a pre-fire NDVI image. The maps formed using ArcMap 10.7 in raster format were transferred to GuidosToolbox software. Landscape analysis for habitat connectivity was carried out using GuidosToolbox (v. 3.0), an integrated platform for the analysis of habitat composition, connectivity and fragmentation, independent from the unique reactions of various species (Vogt & Riitters, 2017).

Table 1. Burn severity levels obtained calculating dNBR, proposed by USGS.

Severity level	**dNBR range (scaled by**	**dNBR range (not scaled)**
Enhance	-500 to - 251	-0.500 to -0.251
Enhance	-250 to -101	-0.250 to -0.101
Unburn	-100 to -99	-0.100 to -0.99
Low	100 to 269	0.100 to 0.269
Moderat	270 to 439	0.270 to 0.439
Moderat	440 to 659	0.440 to 0.659
High	660 to 1300	0.660 to 0.1300

Morphological Spatial Pattern Analysis (MSPA) is a series of specialised mathematical morphologic operators that intend to describe the geometry and connection of image components (Soille & Vogt, 2009; Saura et al., 2011). This method, which is solely based on geometrical concepts, can be implemented on any digital image at any scale and implementation area (Vogt, 2016). MSPA operates with presence/absence data. The morphological structure is categorised into seven classes by the MSPA assessment (Saura et al., 2011). These are: (1) cores (area inside excluding the outside environment); (2) islets (small and distinct areas too small to include cores); (3) loops (areas connected to the same core); (4) bridges (areas connecting different cores); (5) perforations (areas on the periphery of an internal object); (6) edges (areas on the periphery of an external object), and (7) branches (areas with one side connected to an edge, perforation, corridor or loop). Morphological spatial pattern classes are among the significant indicators of habitat fragmentation and loss in forests. For instance, core areas (forests) indicate unfragmented habitat potential for the species living in the forest while edges show the boundaries formed in forests after perforation. Compared to core areas, the edges accommodate invasive species or the species dependent on border areas. Corridors providing connectivity represent the potential mobility of species, the fragmentation of which would cause ecological harm. Branches that can provide connections between core areas are recognised as potential corridors. Certain classes in the interpretation of the analysis can be paired in various combinations. For instance, classes indicating corridors can stand for fragmented or broken connections of three kinds (branches with corridors, branches with an inner corridor and branches formed from the boundary) (Vogt et al., 2007). In total, MSPA segmentation results in 25 classes of characteristics which correspond to the initial foreground area when combined. Since most of the microclimatic and biophysical parameters of forests (temperature, humidity, shade, and radiation) are affected by edge impacts located between 0 and 5 metres from the forest edge, the edge width in the MSPA was adjusted to 3 metres (or 2 pixels) (Ossola et al., 2019). The transformation of morphological classes during the pre-fire and post-fire periods was visualised using Sankey diagrams. The height of each component in the columns representing units in a Sankey diagram is proportional to the relative abundance of morphological classes represented in the study area and the categories were arranged taking into consideration the spatial dimensions (vertical) of all categories, from the smallest to the largest. The thickness of the lines in between columns indicating the units between which change has occurred are important for showing the degree of transformation. A thicker line denotes a higher degree of transformation.

In the final stage of the method, hypsometric values were used to calculate the fragmentation value in percentages. The hypsometric value is scaled to the maximum distance between the foreground (the forest area) and the background (the non-forest area).

The fragmentation of the foreground and background are calculated using the following formulae:

$$\mathrm{frag}_{FG} = \int_0^1 \mathrm{NLCH}_{FG} - \int_0^1 \mathrm{NLCH}_{FGMIN}$$
$$\mathrm{frag}_{BG} = \int_{-1}^0 \mathrm{NLCH}_{BG} - \int_{-1}^1 \mathrm{NLCH}_{BGMIN},$$

where $Frag_{FG}$ in the formula represents foreground fragmentation and $frag_{BG}$ stands for background fragmentation. $NLHC_{FG}$ / $NLHC_{BG}$ is the normalised HE value for the foreground/background class of a specific landscape and $NLHC_{FGMIN}$ / $NLHC_{BGMIN}$ indicates the maximum foreground grouping (minimum fragmentation) for the same foreground ratio.

The degree of fragmentation of a specific image is defined as the area between minimum fragmentation and maximum fragmentation (Kozak et al., 2018). The landscape fragmentation index used in the formation of the normalised HE is represented with values of between 0% and 100% and is used in the calculation of the direction and degree of fragmentation:

$$\mathrm{frag} = f(x) = \left(\frac{A_{FG}}{100} \times \mathrm{frag\_FG}\right) + \left(\frac{A_{BG}}{100} \times \mathrm{frag\_BG}\right),$$

where *frag* indicates fragmentation, A_{FG} indicates foreground, A_{BG} indicates background, *frag_FG* indicates foreground fragmentation and *frag_BG* indicates background fragmentation (Vogt & Riitters, 2017; Kozak et al., 2018).

Results

We find that the NDVI and NBR values in the extent of occurrence (EOO) areas of the species at the study area have significantly changed with the fire. When all the EOO areas were selected from maps belonging to the two spectral indices and compared, we found that the threshold values had dropped. The changes in the EOO areas by burn severity are visualised in Fig. 2. When the burn ratios in different EOO areas are compared, the severity of the fire is observed to have varied. In this study, we find that a total of 751.9 hectares of Lycian salamander habitats were lost due to the mega-fires occurring in summer 2021. Our results suggest that *L. atifi* was the most vulnerable with a loss of 285.84 hectares of habitat area, followed by *L. flavimembris* with 242.54 hectares, and *L. antalyana* with 124.16 hectares. The habitat loss of *L. atifi* within the boundaries of the EOO is concentrated in the northeast and the northwest. The habitat loss of *L. flavimembris* within the boundaries of the EOO is concentrated in the southeast and the northwest. *L. antalyana* suffered habitat loss mainly of low severity within the boundaries of the EOO. *L. fazilae* is the species which suffered the least habitat loss, at 25.83 hectares. The *L. fazilae* habitat suffered low severity fires within the boundaries of the EOO. This is also true of *L. billae* and *L. luschani*. *L. billae* and *L. luschani* suffered habitat losses of 30.83 and 43.40 hectares respectively. The habitats of *L. fazilae*, *L. billae*, and *L. luschani* were thus less affected by the forest fires than those of the other species. The number of EOO areas outside the habitats of *L. flavimembris* and *L. atifi* which suffered high and moderate-high severity fires is low. Table 2 shows that the amount of burn severity in the EOO area for *L. flavimembris* was the most homogenously distributed. *L. flavimembris* and *L. atifi* are the species with the most habitat loss at moderate-high severity. Fig. 3 visualises losses according to burn severity. It shows that *L. antalyana* suffered a 17.40% habitat loss in the EOO area, *L. atifi* a loss of 11.90%, *L. flavimembris* a loss of 9.49%, *L. billae* a loss of 4.44%, *L. fazilae* a loss of 3.10% and *L. luschani* a loss of 1.29%.

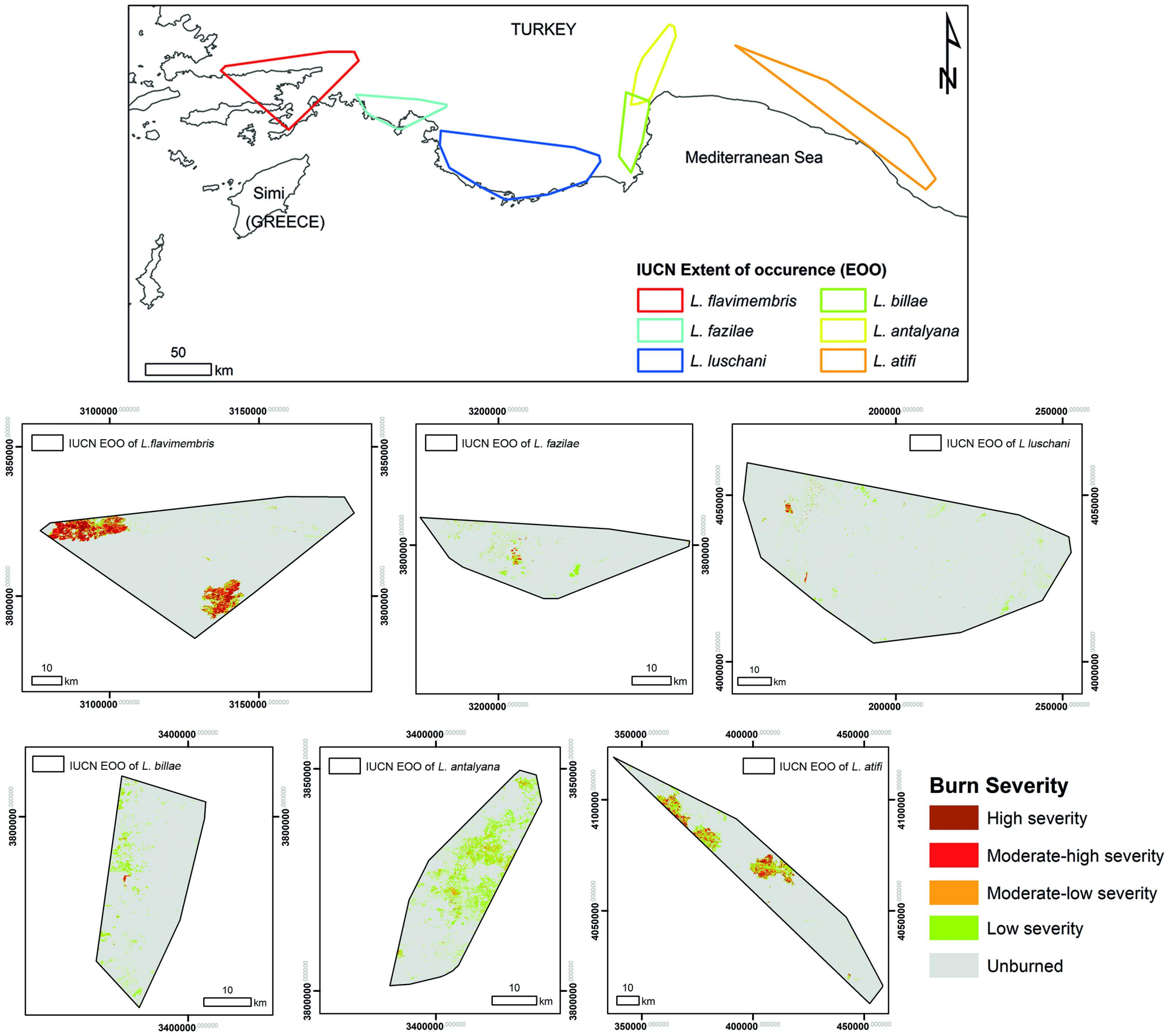

Fig. 2. Burn severity maps for Lycian salamander habitats.

Table 2. Changes in habitats of Lycian salamanders according to burn severity (ha).

Burn severity	***L. antalyana***	***L. atifi***	***L. billae***	***L. fazilae***	***L. flavimembris***	***L. luschani***
Low severity	107.36	145.67	27.61	20.04	56.13	35.74
Moderate-low severity	16.30	79.18	2.55	3.51	54.56	4.27
Moderate-high severity	0.49	40.29	0.60	1.53	67.34	2.82
High severity	0.01	20.69	0.07	0.05	64.52	0.58
Total (ha)	124.16	285.84	30.83	25.13	242.54	43.40

An examination of the MSPA maps displaying morphological classes revealed significant changes in habitat units (Fig. 4). The decrease in the number of core areas demonstrates that all habitats were fragmented after the fire. The loss in core areas overlaps with the EOO areas that suffered habitat losses. For instance, since the habitat loss of *L. atifi* was concentrated in the northeast and the northwest, the loss of core areas was greatest in those regions. Fig. 4 demonstrates the similar spatial results obtained for other species. In addition, the amount of islets, edges, loops, and branches in areas with habitat loss increased and the amount of perforations decreased. Interestingly, in the habitat of *L. atifi*, all morphological classes except for cores increased. However, although the number of bridges in the range of *L. atifi* increased, the increases in the marginal distribution of all classes except cores indicate a very high degree of fragmentation.

When the transformation of morphological classes was examined, a significant decrease was observed in the core areas, which provide spatial connectedness, for all the species, along with an increase in the magnitude of edges, which was taken as the fragmentation indicator (Table 3). In the habitat of *L. atifi*, which suffered a significant loss of habitat, the core areas were seen to have decreased. This was also the case for *L. flavimembris* and *L. luschani*. Bridges providing connectedness were observed to have increased for all species other than *L. fazilae*. This indicates that while existing connections in habitats were fragmented due to the forest fires, potential connections may be formed after the fires.

When we examined the Sankey diagrams, we found that all morphological classes had interchanged after the fire (Fig. 5). Among the morphological classes of *L. flavimembris*, core areas transformed into edges after the fire, whereas for *L. fazilae* core areas transformed into loops. By contrast, a significant proportion of the core areas for *L. luschani* transformed into edges and bridges and some of the branches transformed into islets. Among the morphological classes of *L. billae*, core areas transformed into edges, loops, perforations and bridges. The amount of transformation is similar. While the transformation of the morphological classes for *L. antalyana* resembles that of those for *L. billae*, more transformations from cores to edges occurred in the case of *L. billae*. The greatest marginal transformation in the Sankey diagram was observed for *L. atifi*. While the great majority of the core areas of *L. atifi* transformed into edges, a significant proportion transformed into bridges, islets and branches.

According to the examination of hypsometric values, the fragmentation value of *L. antalyana*, which was 54% before the fire, increased to 58.56% after the fire (Fig. 6). Fragmentation increased from 48.26% to 55.91% in the case of *L. atifi*. There were no significant changes in *L. billae*. The fragmentation value of *L. fazilae* increased slightly from 59.35% to 59.97%. The fragmentation value of *L. flavimembris* increased from 54.76% to 57.74%. In contrast, the fragmentation value of *L. luschani* decreased from 79.47% to 79.04%. In conclusion, the marginal increase in fragmentation before and after the fire was most notable in the habitat of *L. atifi*.

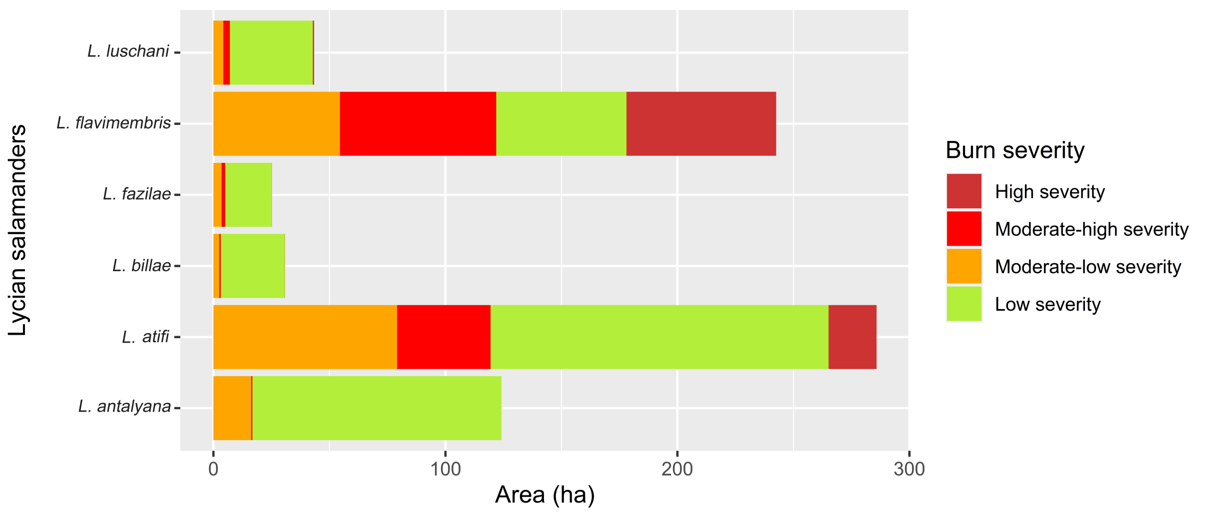

Fig. 3. Comparison of habitats of Lycian salamanders according to burn severity (ha).

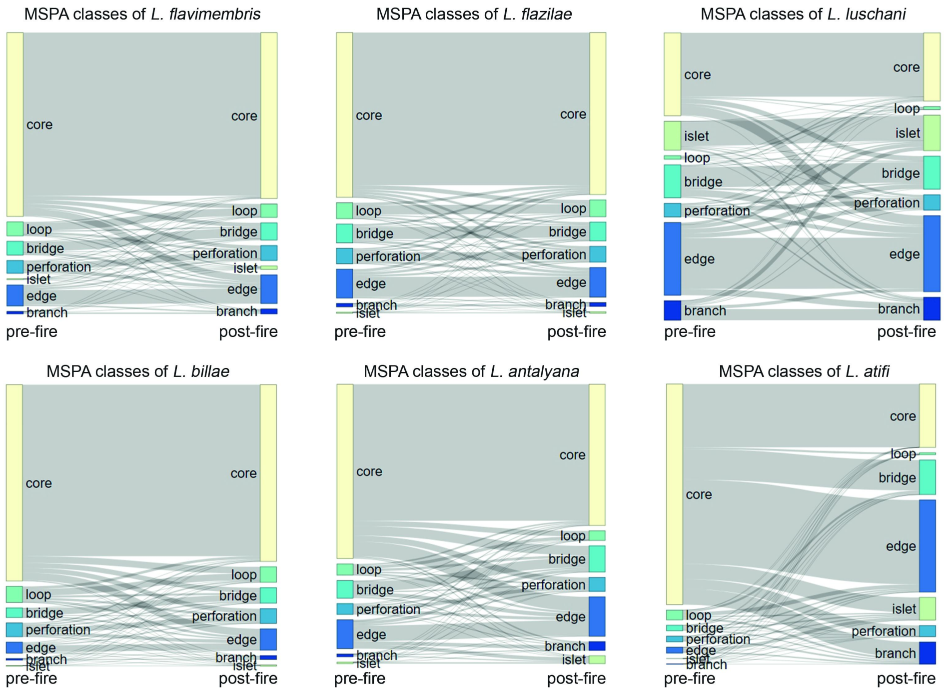

Fig. 4. Change between morphological classes of Lycian salamanders' habitats.

Table 3. Changes in MSPA classes (morphological classes) by species based on pre- and post-fire binary images (km^2).

Species	Core	Islet	Perforation	Edge	Loop	Bridge	Branch
L. flavimembris	-131.37	25.81	-15.85	97.89	9.20	40.13	21.98
L. fazilae	-3.24	0.81	2.43	0.08	0.00	-0.40	0.57
L. luschani	-128.48	64.07	-15.10	22.48	15.43	16.44	24.82
L. billae	-38.68	1.60	-9.10	29.03	0.14	12.08	4.93
L. antalyana	-47.59	16.41	-13.63	31.46	2.07	1.07	10.27
L. atifi	-1215.93	215.51	250.59	589.83	24.75	434.62	195.09

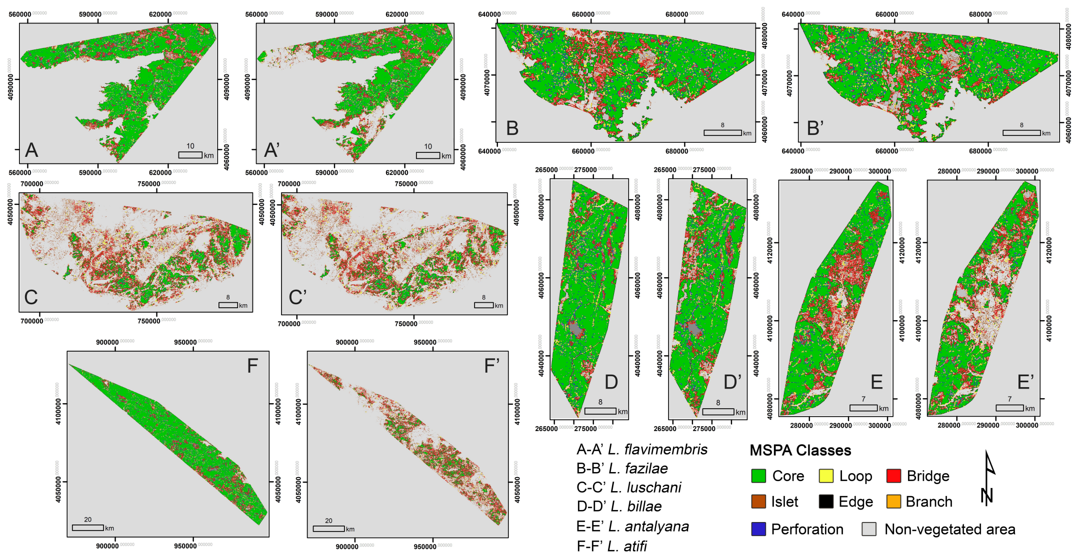

Fig. 5. Sankey diagrams illustrating the transformations of morphological classes of Lycian salamander habitats.

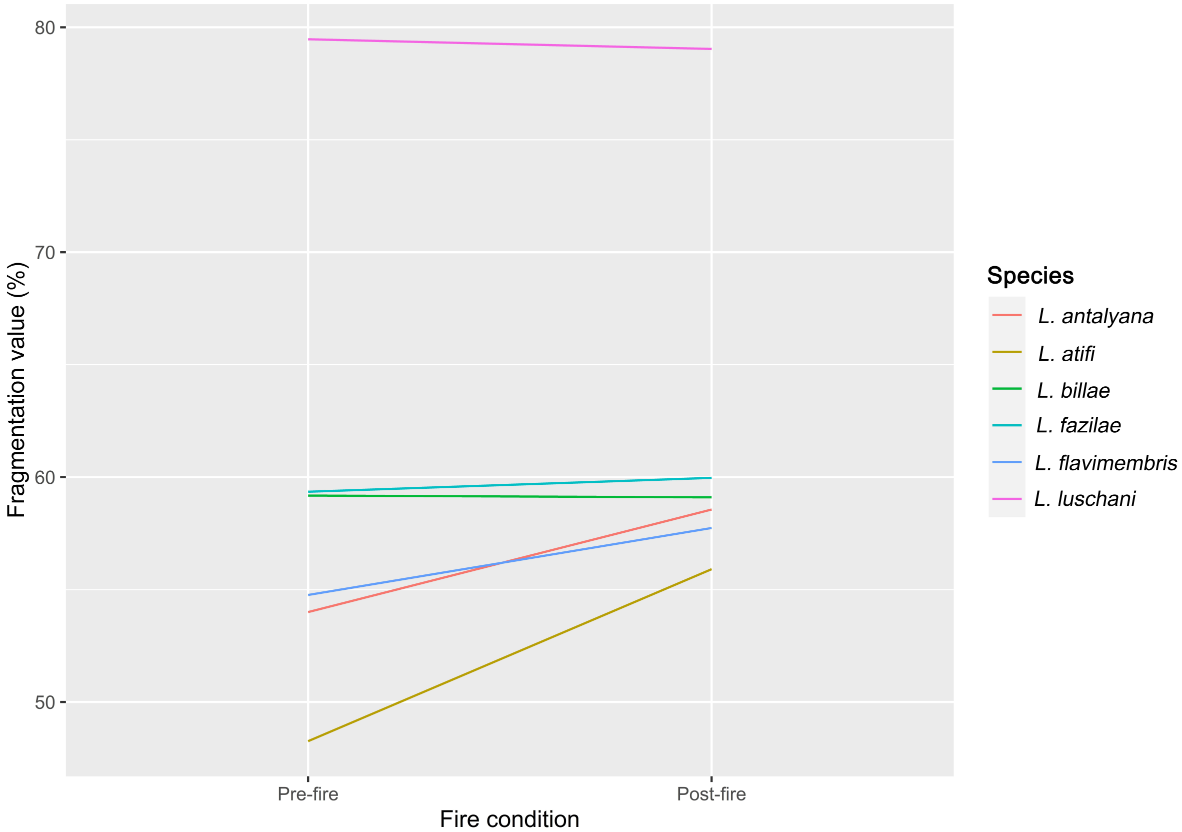

Fig. 6. Change in fragmentation values before and after the fire based on hypsometry values.

Discussion

Seven species of Lycian salamanders, some of which also have various subspecies, are found in the Western Taurus Mountains along the Turkish Mediterranean coastline and the Karpathos Islands of Greece. Lycian salamanders are restricted to karstic limestone areas in arid Mediterranean environments. The species' typical habitat is pine forests and maquis shrublands on north-exposure slopes (Veith et al., 2001). Lycian salamanders live in specific isolated environments and do not display signs of mixing even when living in close proximity (Johannesen et al., 2006). The major threat towards amphibian populations worldwide is habitat loss and habitat fragmentation (Chanson et al., 2008; Sodhi et al., 2008; Hof et al., 2011). The biggest potential threats to Lycian salamanders, which live in naturally isolated environments, are forest fires, urbanisation, overcollection for scientific purposes and habitat loss (Kaska et al., 2009; Başkale et al., 2018; 2019; Arslan et al., 2020).

One of the key natural disruptions to Mediterranean ecosystems, on both small and large scales, is wildfires (Izhaki, 2012). The vertebrate communities of the region have been shaped mostly by fire (Prodon, 1987; 2000). Aside from damaging vegetation, forest fires cause severe habitat and landscape transformations, which affect the dynamics and structures of vertebrates including their population and community levels (Izhaki, 2012). Forest fires reduce the availability of habitats for the species, but the general impact of fires does not have to be negative from the viewpoint of the preservation of biodiversity. This is because they create a heterogeneous landscape through forest patches, and provide open habitats which are significant for the species. Fires affect wildlife directly by causing injury or death and indirectly through their impacts on the quality of habitats and food resources (Smith & Lyon, 2000; Harper et al., 2016). The short-term ecological effects which fires have on animal populations and communities as a result of habitat modification can be even greater than their direct effects (Izhaki, 2012). However, fires occurring in the Mediterranean basin have rather varied effects on biodiversity. These effects depend on the extent, severity and frequency of the fire, the initial condition of the ecosystem and the spatial arrangement and isolation of burned and unburned patches after the fire, as well as various abiotic conditions. In terms of forest management in the Mediterranean, it is of great importance to conserve the landscape mosaic of habitats with different fire histories in order to preserve the high biodiversity of the vertebrate population. The impacts of wildfires on birds and mammals have been documented more extensively than their effects on reptiles; in the case of amphibians, there is only limited data (Izhaki, 2012).

Terrestrial salamanders generally live in shelters, which they leave for foraging if conditions are favourable. Lycian salamanders have been observed to be active from early November to mid-April depending on environmental conditions (Olgun et al., 2001; Gautier et al., 2006; Sparreboom 2014). During this period of activity, the salamanders live in shelters (under stones, in wood or in crevices) and move on the surface when foraging at night (Olgun et al., 2001; Sparreboom, 2014). During the dry season, they avoid the surface and live a secluded life underground, many individuals coming together to share a single shelter (Gautier et al., 2006). Terrestrial salamanders have been sheltered from the direct effects of wildfires due to these underground habits. Plethodontid salamanders retreat underground in wildfires, which most frequently occur during the summer drought (Hossack & Pilliod, 2011). The severity of forest fires consists of two components: intensity and duration. Fire intensity is the speed with which a fire produces thermal energy. Although heat is transferred more rapidly and penetrates deeper in humid soil, the latent heat of vaporisation prevents the soil temperature from exceeding 95°C until the water has evaporated completely (Campbell et al., 1994). The temperature then typically rises to 200-300°C (Franklin et al., 1997). If heavy fuels area present, the temperature on the soil surface may reach 500-700°C (DeBano et al., 1998) and momentary values up to 850°C have been recorded on occasion (DeBano, 2000). The combination of burn and heat transfer produces vertical thermal gradients in the soil. Temperatures at 5 centimetres in mineral soil rarely exceed 150°C and soil 20-30 centimetres from the surface does not generally heat up (DeBano, 2000). The possibility of individuals dying directly from the heat is therefore low. Studies of various species have indicated that direct damage is negligible. The real threats that may impose constraints on the populations of species are increased vulnerability to predators, food scarcity and the microclimatic fluctuations that occur in parallel with the decrease in the vegetation cover ratio. Moreover, the nature and timing of the forestry activities conducted for the recovery of the area after a fire are of great importance.

The Mediterranean flora and fauna have developed considerable defences against forest fires. These species started radiating at least 12.3 and 10.2 mya around the final emergence of the mid-Aegean trench (Veith et al., 2016; 2020). Phylogenetic studies have postulated their intraspecific evolution by the Messinian Salinity Crisis (5.3–5.6 mya) as well as climatic alterations since the Late Pliocene and throughout the Pleistocene (Veith et al., 2016; 2020; Ehl et al., 2019). The demographic and growth model analyses conducted by (Sinsch et al., 2017) show that the deep crevice systems in the calcareous limestone on the shores of the Mediterranean have enabled the Lycian salamanders to survive both the cold winters and the hot dry summers However, forestry activities after the fire may be of greater concern. With the recent wildfires occurring in Turkey, there has been public pressure for the burned areas to be restored. In fact, some civil society organisations have raised donations for the afforestation of burned areas. The forest law of Turkey makes it obligatory to reafforest burned areas within a year. At present, the forestation of these areas is seen as unavoidable in Turkey. Forestation is carried out mechanically by means of deep soil cultivation. In some areas, fragments of rock are cleaned up. Soil cultivation can generally be carried out at all times except the winter months. Unless the salamanders move deeper under the surface for the summer months in the wake of the fire, they will clearly be affected by these cultivation processes.

Logging takes place continuously in the Calabrian pine and scrub forests of Mediterranean Turkey. After the fire, all the trees in some areas are likely to be removed. Coastal giant salamanders (*Dicamptodon tenebrosus*) responded to clear-cut loggings by staying near streams, spending less time in their shelters and decreasing their home range (Johnston & Frid, 2002). Riparian zones are therefore of particular importance in burned forest areas. The higher humidity of these areas enables the vegetation to recover more quickly. These areas are already suitable as salamander habitats. For this reason, they constitute priority zones in the habitat restoration of forest areas.

Logs in burned forest areas that are economically valuable are collected and sold. Considering the magnitude of the area affected by this mega fire, removing the logs will require heavy-duty machinery, which brings with it the risk of greater soil compaction during the removal of logs (Grialou et al., 2000). The use of heavy-duty machinery should be prohibited in areas with high potential for habitat suitability as identified by scientific studies. Operations of this kind would not only affect salamanders but also the endemic fossorial *Ophiomorus kardesi* (Kornilios et al., 2018), as well as other amphibians and reptiles living in these areas.

Clearing away the plant residue covering the soil surface could cause problems. (Morneault et al., 2004) argue that terrestrial salamanders will be more affected by the disappearance of the humid microclimate provided by woody residues than by timber logging. Moreover, surface cover could permit salamanders to move more freely without being attacked by predators. For instance, *Ambystoma opacum* moves for relatively short distances until it finds shelter (Graeter et al., 2008). Of course, the surface cover does not consist only of vegetation. The distribution areas of the Lycian salamanders consist of deep crevices and calcareous rocks with rough surfaces. This constitutes an advantage for the salamanders.

Another factor to be considered during the habitat restoration of burned areas is timing of operations. The only period when Lycian salamanders are active on the soil surface is winter (Franzen, 2008). This indicates that this period is important for the survival of the population (Sinsch et al., 2017). Because of this, the area should be screened extensively before each and every operation to be carried out in the winter and precautions should be developed and applied instantly depending on the populations identified.

Populations with restricted distribution, including Lycian salamanders, are more prone to extinction unless their distribution is secured by ecological connections (Bani et al., 2015). The degree of isolation between the populations can be determined by ecological and geographical distances. Karstic limestone bedrock, rivers, and rock formations have been seen as the greatest obstacles to the gene flow of Lycian salamander populations (Klewen, 1991). On the contrary, Veith et al. (2020) claim that karstic limestone could even connect adjacent populations, making the distribution through river systems easier. Nevertheless, due to their environmental tolerance limits and limited dispersal capacities, the species were unable to spread during the climate changes in the history of the Mediterranean, where southern Anatolian environments are even more suitable for the survival of Lycian salamanders today (Veith et al., 2016; 2020). All the Lycian salamander species are similar in terms of their ecological niches and selection of habitats, and there are no significant differences between them (Franzen, 2008).

Spatial planners need easy-to-use and effective alternative planning tools to minimise habitat loss, prevent fragmentation and strengthen ecological connectedness (Babí Almenar et al., 2019). To preserve ecological assets and ensure their balanced use, preservation methods should be researched, analysed and planned. The implementation of planning decisions should be inspected, and interventions should be made if necessary. Throughout this process, there is a need to produce base maps to direct planning decisions. In this context, it is important to monitor the transformations in ecological connectedness units and to assess the changes in the importance of connectedness as part of conservation planning.

There is a strong correlation between measurements of habitat configuration and habitat volumes (Fahrig, 2003). Landscape ecologists argue that forests, which cover large spaces in natural ecosystems, may contain a particularly large variety of habitat types. A greater diversity of species may emerge in large and continuous forests, since larger areas generally possess a regional pool of species. For this reason, most studies argue that forest fragmentation leads to a reduction in biodiversity (Jackson & Fahrig, 2013). Although this study does not focus on the impacts of direct fragmentation on biodiversity, fragmentation occurring over time is thought to damage habitat areas. Significant fluctuations may occur in the presence of individual species and/or their transformation as a result of the decrease in forest areas under the impact of fragmentation. Additionally, a significant edge impact may occur, and the survival rate of trees in forest areas that are opened up in mountain ecosystems may decrease due to increased exposure to wind (Laurance, 2008; Kettle & Koh, 2014). For this reason, species may be lost in forest areas affected by fragmentation or the suitability of these areas as habitats for species may change. Identifying positive and negative transformations may lead to practical solutions for conservation management in landscape planning.

Identifying the species most affected by fragmentation over time could be useful for understanding the condition and dynamics of forest ecosystems, habitat conditions and ecosystem functions (Hermosilla et al., 2019). This study has focused on the short-term transformations over time on Lycian salamander habitats as a result of forest fires. Due to the lack of data on habitats after the fire, it has not been possible to make an assessment of functional connectedness. Subsequent studies may develop both structural and functional connectedness models to assess the connectedness of green spaces more comprehensively. To this end, we recommend adopting data collection methods that are based on field observations and that make it possible to understand the mobility of the species in the landscape.

The available research suggests that data on land cover or pixel sizes on different spatial scales can lead to different results during fragmentation analysis. For instance, Riitters et al. (2002) emphasise that smaller pixel sizes or fragmentation analysis conducted with a higher resolution database will not show greater fragmentation than larger pixels or lower resolution data. However, other studies indicate that higher resolution data can display more detailed anthropogenic fragmentation (Shen et al., 2020).

The capacities of forests and other natural habitats to sustain biodiversity and maintain ecosystem services will depend on the total amount and quality of the fragmented habitats, on their connectivity levels and on how they are affected by climate change. In order to explain and estimate the long-term environmental impacts of transformations in habitats in forest areas, more research needs to be done using a variety of scenarios. Subsequent studies should address various ecosystems and spatial connectedness as a whole, and present experimental efforts within a holistic approach (Hansen et al., 2013).

For the sustainable planning and management of areas with strong natural landscape characteristics, it is crucial to analyse the spatial transformations of habitat units over time using the latest data and methods so as to obtain accurate and useful results. Taking into account the interaction between the species and the landscape, it has been argued that the continuity of local populations must be ensured in core areas consisting of forests. Analysing maps of the ecological connectedness provided by green spaces may prove beneficial in identifying priorities for the conservation of areas with high rates of fragmentation.

This study underlines that the areas of burned forests and the burn severity can be analysed using remote sensing methods. In many studies, NBR and dNBR have performed well in identifying burned areas and analysing burn severity (French et al., 2008; Sabuncu & Özener 2019; Roy et al., 2019). However, since we could not acquire information on ignition points, we were unable to validate the classification of the Landsat-8 images. This may be considered a limitation of the study.

Conclusions

The present study has assessed the fragmentation in the distribution areas of six species due to forest fires using Landsat 8 satellite images and remote sensing and pattern analysis methods. Habitat loss increased substantially as a result of the fire, depending on burn severity. The changes in morphological classes showed that while the core areas in habitats decreased the formation of new bridges hints at the possibility of a recovery of spatial connectedness. However, since the formation of new bridges was not able to prevent an increase in other morphological classes, it is included that the fragmented areas need to be restored urgently. For these reasons, the survival of the salamander populations in the fragmented landscapes depends on the preservation or restoration of the functional (ecological) connectedness between the remaining habitat areas, which needs to be assured by the presence of ecological corridors.

Large-scale forest fires are one of the environmental disruptions that negatively affect habitats, particularly in Mediterranean Turkey. This study therefore emphasises that landscape administrators and environmental policy-makers need to come up with appropriate strategies that will prevent forest fires from occurring. Monitoring the spatial structure of forest areas will be decisive in whether these strategies achieve their aims. This process will succeed if effective, well-planned and well-coordinated measures are taken that also contribute to forest ecosystem management. For the purposes of the planning activities to be carried out in this context, it is proposed that fire risk maps should be developed using satellite image and pattern analysis techniques, and that these maps should be incorporated into environmental monitoring activities.

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