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Integrating Citizen Science and Multispectral Satellite Data for Multiscale Habitat Management

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Abstract

Habitat management is necessary for the conservation of threatened species, yet best practices in fragmented human-dominated landscapes have remained difficult to generalise. We show that multi-scale vegetation management decisions in heathlands can be supported by integrating opportunistic citizen science data and multispectral satellite data.

Opportunistic observations were gathered from ten typical, mostly threatened animal species of dry heathlands in Flanders as point records with specified precision. We considered vegetation structure at the local scale, quantified by image texture within 0.25 hectares derived from multispectral satellite data, and heathland heterogeneity at the habitat scale, quantified by the diversity in heathland vegetation communities within 50 hectares. Additionally, locations inside heathlands were attributed to an open, closed or anthropogenic landscape context. Point process models were used to test the impact of heathland size, vegetation structure and heathland heterogeneity on the habitat suitability of the studied species.

We found that (1) heathland vegetation management can benefit habitat suitability in fragmented heathlands, but with a different approach for local management of vegetation structure in small versus large heathlands (e.g. due to micro-fragmentation effects), (2) the landscape induces positive and negative edge effects (e.g. due to a high versus low resource availability), especially in small heathlands and (3) habitat suitability is driven by both vegetation structure and heathland heterogeneity but with different relative importance for birds, butterflies and grasshoppers (e.g. due to differences in mobility).

Keywords: habitat suitability, image texture, point process models, remote sensing

1. Introduction

Dry heathlands are human-shaped habitats prioritised in Annex I of the European Habitats Directive (92/43/EEC). They can provide a variety of ecosystem services, such as food and water supply, landscape and biodiversity conservation, carbon sequestration and aesthetic/recreational value. They are, however, threatened by land conversion, privatisation, recreation and soil eutrophication from intensive agriculture that causes moss, grass and tree encroachment (Fagúndez, 2013; Webb, 1998). These pressures have led to the fragmentation and reduced habitat quality of heathlands, and an ever-increasing proportion of heathland fauna appearing on national red lists (Maes et al., 2019a). Conservation of species that rely on habitats under anthropogenic pressures remains challenging and is in strong need of evidence-based action plans (Maes et al., 2022; Olmeda et al., 2020). In European dry heathlands, conservation management is traditionally designed from a flora perspective with a focus on preserving typical successional heathland vegetation (De Blust, 2022; Webb, 1998). Typically, management schemes are designed to prevent nutrient accumulation and natural succession to forest, for example by sod-cutting, burning or grazing (De Blust, 2022; Fagúndez, 2013). It has become generally accepted, however, that heathland fauna profits from management that includes exposure of bare soil, diversifies vegetation communities and increases structural complexity (Byriel et al., 2023; De Blust, 2022; de Vries et al., 2021; Schirmel et al., 2011; van den Berg et al., 2001), yet evidence-based action plans in conservation policy remain scarce.

The quality of a habitat, in terms of its suitability for animal species of conservation interest, depends on its environmental characteristics and spatial context. First, habitat suitability is usually higher when a broad range of environmental resources is available (for example for taking shelter, nesting and foraging) (MacArthur and Wilson, 1967; Tews et al., 2004). Second, habitat size generally impacts habitat suitability positively (e.g. Fahrig, 2003; Milanesi et al., 2017; Rutten et al., 2019; van den Berg et al., 2001), although the potentially high suitability of small and isolated patches should not be neglected (Wintle et al., 2019). Third, habitat suitability is also impacted by the landscape context, which can provide opportunities for habitat connectivity (Gibson et al., 2004; Haddad and Baum, 1999) or induce positive or negative edge effects (Dupont and Overgaard Nielsen, 2006; Fagúndez, 2013; Neilan et al., 2019; Pfeifer et al., 2017). While the impact of edge effects on heathland vegetation is spatially confined (e.g. eutrophication effects on vegetation and soil were detected up to ca. 8 metres into heathland patches according to Piessens et al. (2006)), the impact on heathland fauna might reach further (e.g. Pfeifer et al., 2017).

Vegetation management can diversify environmental resources for animal species and thereby increase habitat suitability. Heathland vegetation management can be implemented at different spatial scales, of which we distinguish two in this study. At the larger habitat scale, we define *heathland heterogeneity* as the horizontal diversity of heathland-associated habitat subtypes (i.e. vegetation communities) such as wet and dry heathlands, peatlands and/or Nardus grasslands. At the smaller local scale, we define *vegetation structure* as the variation in vegetation height within a heathland (i.e. bare ground, heathland of different age classes, the presence of shrubs and/or trees; Bergen et al., 2009). Both scales are crucial for determining species' habitats (Bergen et al., 2009; Randin et al., 2020) where, firstly, a high habitat heterogeneity leads to higher habitat suitability for species that need complementary resources (see reviews of Stein et al., 2014 and Tews et al., 2004). Secondly, vegetation structure has been shown to impact the habitat suitability of birds in forests (Farrell et al., 2013; Goetz et al., 2010; Graf et al., 2009; Huber et al., 2016; Seavy et al., 2009) and grasslands (Bellis et al., 2008), butterflies in grasslands and woodlands (de Vries et al., 2021) and lizards in a river valley (Sillero and Gonçalves-Seco, 2014).

Measures of heathland heterogeneity and the landscape context can easily be obtained by a landscape analysis, for example by using landscape metrics based on regional land use maps (Gustafson, 1998; Hesselbarth et al., 2019). Measures of vegetation structure in low-stature habitats such as heathlands are commonly obtained through airborne Light Detection and Ranging (LiDAR) sensors, because of the fine-resolution data they can obtain across large spatial extents (de Vries et al., 2021; Moudrý et al., 2022; Wehr and Lohr, 1999). LiDAR can deliver accurate measures of vegetation height, although expensive data acquisition and limited temporal coverage reduce its application potential for habitat suitability modelling (Moudrý et al., 2022). For instance, there can be an undesired temporal mismatch between LiDAR-derived explanatory variables and species occurrence data (Randin et al., 2020). Furthermore, when LiDAR images are taken in the leaf-off season, they fail to accurately capture the vegetation structure of low-stature habitats (e.g. in grasslands; de Vries et al., 2021).

For habitat suitability modelling, alternatives are therefore being explored to remotely quantify vegetation structure across large spatial and temporal extents, such as multispectral data from spaceborne sensors. Multispectral Sentinel-2 data are especially interesting for capturing the temporal dynamics of vegetation structure over large areas as they are freely accessible, with four bands (RGB-NIR) at 10-metre resolutions every 5 to 10 days since April 2017. Both the growing availability of multispectral Sentinel-2 data and the increasing quantity and density of species occurrence data through large citizen science initiatives, such as *waarnemingen.be* in Flanders (https://www.waarnemingen.be) and observation.org (https://observation.org/) or iNaturalist (https://www.inaturalist.org/) worldwide, facilitate the use of fine-grained habitat suitability models (Milanesi et al., 2017; Randin et al., 2020) such as point process models (PPMs) (Renner et al., 2015). The strength of PPMs is that environmental data can be extracted for each point observation of a species' presence at fine spatial resolutions, which allows to assess fine-scaled drivers of habitat suitability (also see Appendix A).

When LiDAR data are not available, a good alternative to measuring vegetation structure across large spatial extents is satellite image texture (Wood et al., 2012). Image texture quantifies the spatial heterogeneity in pixel tone values of a certain image area (Haralick, 1979) and can therefore be used to quantify the spatial heterogeneity in images of vegetation cover (Wood et al., 2012). For applications in habitat suitability modelling, a recent study proposed the use of Sentinel-2-derived image texture based on the Enhanced Vegetation Index (EVI; Liu and Huete, 1995) in low-stature habitats, i.e. grasslands and shrublands (Farwell et al., 2021).

Here, we test the possibility of integrating opportunistic citizen science data and multispectral satellite data to support multiscale management decisions for the conservation of animal species in anthropogenic regions (Maes et al., 2022). More specifically, we will analyse whether the habitat suitability of dry-heathland species across different taxonomic groups is driven by vegetation structure and/or heathland heterogeneity and whether this relationship depends on the heathland size and landscape context. We hypothesize that heathland management can benefit habitat suitability for species of conservation interest, even in small heathlands (Gábor et al., 2022; Wintle et al., 2019), that it should consider the landscape matrix due to positive and negative edge effects (Fahrig, 2003) and that it requires an integrated multispecies approach (Bonari et al., 2017; Maes and Van Dyck, 2005).

2. Methods

2.1. Study area

The study region was the Campine region in Flanders in the northeast of Belgium (Fig. 1a), holding about 13,000 hectares of heathland (De Saeger et al., 2020) and characterised by sandy soils (Couvreur et al., 2004). We limited our study area to heathland patches with more than 40 per cent classified heathland on the 2020 Biological Valuation Map (BVM) (De Saeger et al., 2020) which is a database for land cover in Flanders that includes a map of habitat classes (De Saeger et al., 2017). We omitted three military domains (Fig. 1a), because of a strong negative observation bias due to their inaccessibility, and patches with urban elements.

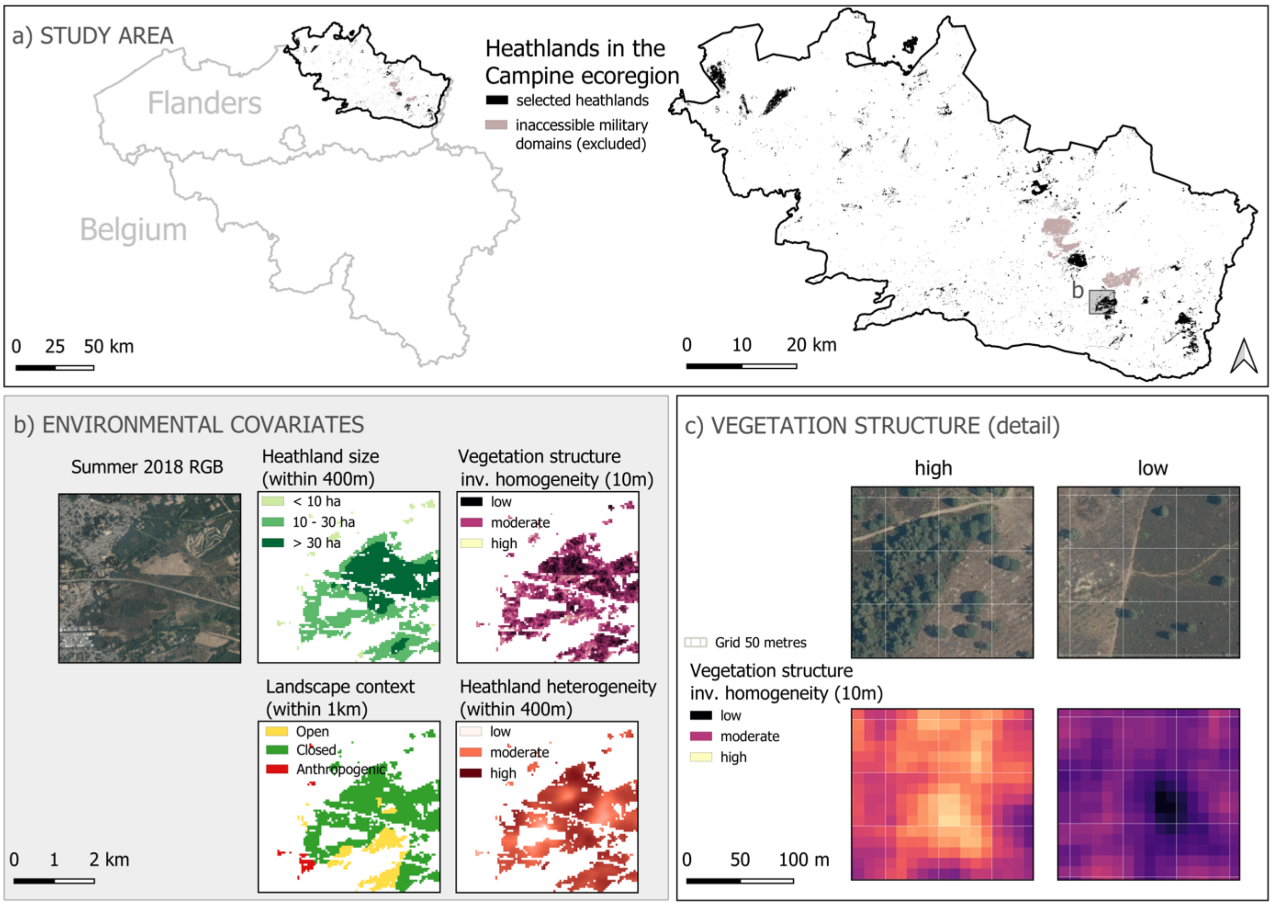

Fig. 1: a) Study area and studied heathlands (inaccessible military domains were excluded); b) Environmental covariates used to predict the relative habitat suitability of dry-heathland species of conservation interest at an example location. The landscape context was the dominant surrounding land cover class in a one-kilometre radius around points on a regular grid of 50 metres. Heathland size and heathland heterogeneity were calculated as the mean heathland size and the Shannon diversity in heathland subtypes (such as dry and wet heathlands and heathlands with and without trees), respectively, within a 400-metre radius around points on a regular grid of 50 metres. Vegetation structure is the inverse of the homogeneity (a gray-level co-occurrence matrix (GLCM) second-order texture metric) in the Enhanced Vegetation Index, calculated at a resolution of 10 metres, supplemented with the average homogeneity in a 50-metre radius around points on a regular grid of 50 metres in the patch edges with missing values; c) Example locations with low and high vegetation structure (i.e. spatial variation in vegetation height). The location with low structure is characterised by more uniform age-classed dry heather vegetation communities (Calluna – Genista) with few or no trees or shrubs. The location with high structure is characterised (from left to right) by plantings of Scots pine (Pinus sylvestris L.) with undergrowth of shrubs and trees, a woody edge of broom thicket (Cytisus scoparius L.) and different age classes of dry heather

vegetation (Calluna vulgaris L.) with shrub or tree stands (vegetation cover as described on the Biological Valuation Map version 2020 (De Saeger et al., 2020)).

2.2. Species observations

We considered dry-heathland fauna of conservation interest in Flanders, meaning that they are either species of regional conservation interest (Annex II or IV of the Habitats Directive (92/43/EEC) or Annex I of the Birds Directive (79/409/EEC)) (Paelinckx et al., 2009), Flemish Priority Species (De Knijf et al., 2014; Herremans et al., 2014) or habitat-specific species (Habitats Directive habitat types 2310, 2330, 4030) (De Knijf and Paelinckx, 2013) (Tables 2 and B.1). Critically endangered species were excluded (e.g. Northern wheatear *Oenanthe Oenanthe*), mostly because of the low number of observations. Observations from eighteen species from four taxonomic groups (i.e. four birds, five butterflies, seven grasshoppers and two reptiles) were extracted from the data portal *waarnemingen.be* (Herremans et al., 2018; https://www.waarnemingen.be). They were point observations with specified geographical precision for the study region and study period 2017-2021. Only the months from April to August were considered as this period provided a good overlap between the growing season in Flanders and the reproductive seasons for the focal species. The data was cleansed, checking for wrong coordinates, removing incorrect observations and keeping only observations with a precision below 50 metres.

To construct the model training sets, we extracted opportunistic/unstructured records and first applied data quality filtering according to previous recommendations made by Van Eupen et al. (2022). Data verified as correct were retained based on the taxonomic group, range size and relative body size. Second, we applied spatial thinning at 50 metres per observation date to reduce the impact of duplicates (i.e. observations from an individual at a similar location on the same date). Table B.1 shows the number of presences remaining after data cleansing, data quality filtering and spatial thinning.

2.3. Model covariates

We used existing maps, satellite imagery and species occurrence data from *waarnemingen.be* to compile the model covariates, i.e. four environmental covariates (landscape context, heathland size, heathland heterogeneity and vegetation structure) and two sampling bias covariates (accessibility and search effort) (Table 1; sections 2.3.1 and 2.3.2). Most model covariates (all except vegetation structure) were rasterised at a resolution of 50 metres by applying calculations (i.e. summary statistics, landscape metrics, vector lengths) in a buffer area with varying radii around each point at a regular grid of 50 x 50 metres, further called ‘dummy points’. Per species, all model covariates were tested for multicollinearity by extracting their values at all training presence locations and calculating variance inflation factors (VIFs) and Pearson correlations in the R package ‘fuzzySim’ version 4.3 (Barbosa, 2015). All calculations were performed in R version 4.2.1 (R Core Team, 2022) and QGIS version 3.16.9.

Table 1: Methods applied to obtain the model covariates that were used to predict the relative habitat suitability of dry-heathland species of conservation interest. Dummy points are points at a regular grid of 50 x 50 metres throughout the study area. BVM = Biological Valuation Map, rasterised at 5 metres; EVI = Enhanced Vegetation Index; GLCM = gray-level co-occurrence matrix; [1] *De Saeger et al. (2020);* [2] *retrieved from Google Earth Engine;* [3] *retrieved from https://land.copernicus.eu/pan-european/corine-land-cover/clc2018;* [4] *retrieved from https://www.geopunt.be/;* [5] *Herremans et al. (2018).*

Covariate	*Source*	*Calculation (per pixel)*	*Scale*	*Res.*	*min-max*
ENVIRONMENTAL COVARIATES					
Landscape context	*Flanders*: BVM version 2020 [1] *Outside Flanders:* CORINE version 2018 [3]	Formed 3 land use classes: closed (forest) open (other semi-natural) anthropogenic (urban and agricultural)	Dominant class within one kilometre around each dummy point	50 m	NA (factor)
Heathland size (hectares)	BVM version 2020 [1]	Percentage of heathland (converted to hectares)	Mean within 400 metres (≈ 50 ha) around each dummy point	50 m	0.22 - 49.85
Heathland heterogeneity	BVM version 2020 [1]	4 heathland subtypes (each with/without trees or shrubs) - dry heathlands - wet heathlands - peatlands - Nardus grasslands	Shannon diversity index within 400 metres (≈ 50 ha) around each dummy point	50 m	0.00 - 1.71

Covariate	*Source*	*Calculation (per pixel)*	*Scale*	*Res.*	*min-max*
Vegetation structure	Sentinel-2A images [2] April to August 2017-2021 Heathlands (> 40% heathland) and semi-natural edges (> 10% heathland)	Masked clouds, snow/ice and unreliable pixels. Calculated EVI (kept values between 0.1 and 1). Averaged annual median composites. GLCM 2nd order texture: homogeneity (inverse).	5 x 5 moving window (≈ 0.25 ha) in steps of 10 metres Pixels with missing values in moving window: average homogeneity (inverse) within 50 metres	10 m	0.06 - 0.97
SAMPLING BIAS COVARIATES					
Accessibility (km road/km^2)	Wegenregister version 2.0 [4]	Length of road segments	within 100 metres (≈ 3.14 ha) around each dummy point	50 m	0.00 - 0.05
Search effort (n° species)	*waarnemingen.be* [5] April to August 2017-2021	The annual average number of species observed within the considered taxonomic group	within 100 metres (≈ 3.14 ha) around each dummy point	50 m	0.0 - 37.4

2.3.1. Environmental covariates

We chose to include only measures of vegetation structure and habitat composition, although we acknowledge that including measures of soil water, such as the topographic wetness index (Besnard et al., 2015; Moore et al., 1993), or soil biochemistry, such as nitrogen and phosphorus content (Vogels et al., 2017), might have led to additional insights. We motivate the choice of our covariates by the objective of the study (i.e. to illustrate how integrating citizen science and multispectral satellite data can support multiscale heathland vegetation management) and the absence of multicollinearity (Table B.2). Soil water, for example, might be correlated with both heathland heterogeneity and vegetation structure, as soil moisture impacts the composition of vegetation communities and the presence and growth of certain plant species (Schellenberg and Bergmeier, 2020).

a) Landscape context

Landscape context, but also heathland size (section 2.3.1.b) and heathland heterogeneity (section 2.3.1.c) were calculated based on the Biological Valuation Map (BVM) as it includes a detailed classification of habitat types and a classification of land cover in Flanders (De Saeger et al., 2017). To describe the landscape context, we categorised the land cover into three classes: closed (forest), open (all other semi-natural land covers) and anthropogenic (urban and agricultural) land use. The dominant class in a one-kilometre buffer around each dummy point was taken as the landscape context (Fig. 1b). Land cover within the Campine region was taken from the BVM, while at the borders of Flanders, we used the CORINE land cover map (https://land.copernicus.eu/pan-european/corine-land-cover/clc2018).

b) Heathland size

To quantify heathland size, we attributed the percentage of heathland associated with each 5-metre pixel in the rasterized BVM following the distribution key of the different habitat units per patch (De Saeger et al., 2020). Consequently, we calculated the mean percentage of heathland in a 400-metre radius (≈ 50 hectares) around each dummy point (Fig. 1b). We chose 400 metres as the maximum radius for all species to facilitate comparability among results. Models were run with continuous heathland size as a covariate to assess its impact on habitat suitability. However, we also categorised heathland size into three classes in the results section for the dual purpose of simplifying the presentation of the results and formulating tangible recommendations. We distinguish between (1) small patches (≤ 10 hectares), i.e. mostly small and isolated patches with an occasional heathland patch edge largely surrounded by other land use, (2) intermediate patches/patch edges (10-30 hectares), i.e. mostly edges of large heathland patches with an occasional medium-sized patch, and (3) large patches (> 30 hectares), i.e. core areas of large heathland patches (Fig. 1b).

c) Heathland heterogeneity

Heathland heterogeneity was quantified by the Shannon Diversity Index (shdi) in the R package ‘landscapemetrics’ version 1.5.4 (Hesselbarth et al., 2019), applied to four sub-types of heathland as classified in the BVM version 2020 (De Saeger et al., 2020): dry heathland, wet heathland, peat and Nardus grasslands. We also distinguished subtypes with and without trees or shrubs. The BVM was rasterised at 5 metres and the shdi was calculated in a 400-metre radius (≈ 50 hectares) around each dummy point (Fig. 1b). The BVM is a vector layer but was rasterized because the ‘landscapemetrics’ package takes raster layers as input.

d) Vegetation structure

We used Sentinel-2A imagery to quantify vegetation structure as this satellite has been delivering multispectral data across large spatial extents since April 2017 at a high spatial and temporal resolution (10 x 10 metres every 5 to 10 days for Flanders). Vegetation structure was quantified by calculating the homogeneity, a second-order texture measure for image smoothness (Haralick, 1979; Haralick et al., 1973), of a Sentinel-2 EVI (Enhanced Vegetation Index) composite (Liu and Huete, 1995) (Fig. 1b). When calculating second-order texture measures, the spatial configuration of pixel values is taken into account by first constructing a gray-level co-occurrence matrix (GLCM; Haralick et al., 1973). Second-order homogeneity characterizes mainly vertical complexity with ancillary information on horizontal plant diversity and was suggested to sufficiently capture vegetation structure relevant to species' habitat suitability (Farwell et al., 2021). Fig. 1c shows examples of heathland locations with low structure (i.e. a mostly uniform vegetation cover) and high structure (i.e. a high variability in vegetation height).

For each 10-metre pixel in the Campine region, annual median EVI composites from April to August in the study period 2017-2021 were obtained from the near-infrared, blue and red band of the image collection “Sentinel-2 MSI: MultiSpectral Instrument, Level 2A” in Google Earth Engine. Before calculating the EVI, pixels with scene classification labels 1 to 3 and 8 to 11 were omitted (i.e. unreliable pixels, clouds and snow/ice). The annual EVI values were averaged, excluding values below 0.1 and above 1 as they mostly indicated buildings, paved soils or solar panels. Homogeneity was calculated using the R package ‘glcm’ version 1.6.5 (https://cran.r-project.org/web/packages/glcm/) with a kernel size of 5 (i.e. a moving window of 5 x 5 pixels or 50 x 50 metres). Vegetation structure was calculated in steps of 10 metres and the inverse of homogeneity was taken as low values indicated a high vegetation structure and vice versa.

We adapted our approach to increase the availability of pixels available for modelling despite the large number of edges in our study area. Since the study area was not a spatially continuous patch, edges were abundantly present inducing one or more missing EVI values in the moving windows used to calculate homogeneity. To reduce the impact of these edge effects and hence increase the number of raster pixels with covariate values for vegetation structure, we took three actions. First, for texture calculations, we included the EVI values from semi-natural edges (i.e. connected patches of semi-natural habitats of which at least 10% was identified as heathland). Second, we chose a small kernel size to reduce the chance of missing values for texture calculations. Third, we calculated the average homogeneity in a 50-metre radius around each dummy point in the patch edges with missing values and added this information to the raster layer for vegetation structure.

2.3.2. Sampling bias covariates

In a point process setting, it is common to include additional covariates that can accommodate sampling bias instead of modifying the background (i.e. the quadrature scheme) (Renner et al., 2015), as opposed to, for example, a target group background selection in Maxent (Phillips et al., 2009). We added one accessibility covariate: road density (km road per square km); and one search effort covariate: the annual average number of species observed within the considered taxonomic group in the study period (extracted from *waarnemingen.be*). Both were calculated in a 100-metre radius around each dummy point. Accessibility accounted for the impact of high observation density around roads (both paved and unpaved) while search effort accounted for the impact of observer activity.

2.4. Species distribution model

We ran Gibbs point process models with a Geyer saturation process (Baddeley et al., 2015) per species and per landscape context to study the impact of heathland size, vegetation structure and heathland heterogeneity on the habitat suitability of dry-heathland fauna in different landscape contexts. Goodness-of-fit was evaluated with a Diggle-Cressie-Loosmore-Ford (DCLF) test (Baddeley et al., 2014) and predictive performance was assessed in a spatial block cross-validation using the R package 'blockCV' version 3.1-1 (Valavi et al., 2019). Appendix A motivates the selection of the SDM method (section A.1), specifies the model parameters and settings used in this study (section A.2) and elaborates on the methods used for assessing model performance (section A.3).

We encountered some model fitting problems in exploratory analyses and set a threshold of 60 presences to avoid poorly fitted or invalid models. Eight species and four models in the anthropogenic landscape context were, therefore, omitted for further analysis (see Table B.1). We finally kept ten species with valid models in at least two landscape contexts (Table 2).

Table 2: List of selected species with their Red List Status in Flanders (LC = Least Concern, NT = Near Threatened, EN = Endangered) (Devos et al., 2016; Maes et al., 2017a, 2021), Conservation Interest (BD = Birds Directive, FPS = Flemish Priority Species, HSS = Habitat Specific Species with Habitats Directive Annex I habitat types) (De Knijf et al., 2014; De Knijf and Paelinckx, 2013; Herremans et al., 2014; Paelinckx et al., 2009), and species occurrence in the different landscape contexts (number of presences, after data quality filtering and after spatial thinning per observation date at 50 metres, and average intensity of the point process expressed as the number of presence points per hectare).

Species	**English name**	**Red List status in Flanders**	**Conservation Interest**	**Number of presences** *and average intensity (n° points per hectare)*		
				Open	Closed	Anthr.
BIRDS						
Anthus trivialis	Tree Pipit	NT	HSS ^{2310}	907 *0.435*	2638 *0.637*	137 *0.100*
Caprimulgus europaeus	European Nightjar	NT	BD Annex I HSS ^{4030}	151 *0.072*	462 *0.110*	-
Lullula arborea	Woodlark	NT	BD Annex I HSS $^{2310,\ 4030}$	492 *0.236*	1213 *0.288*	80 *0.059*
Saxicola rubicola	European Stonechat	LC	HSS $^{2310,\ 4030}$	935 *0.448*	1542 *0.366*	130 *0.095*
BUTTERFLIES						
Callophrys rubi	Green Hairstreak	EN	HSS $^{2310,\ 4030}$	265 *0.127*	321 *0.076*	-
Hipparchia semele	Grayling	EN	FPS	302	330	485

Species	English name	Red List status in Flanders	Conservation Interest	Number of presences *and average intensity (n° points per hectare)*		
				Open	Closed	Anthr.
			HSS $^{2310, 2330, 4030}$	*0.145*	*0.078*	*0.355*
Plebejus argus	Silver-studded Blue	EN	HSS ^{4030}	621 *0.298*	483 *0.115*	-
GRASSHOPPERS						
Gryllus campestris	Field Cricket	EN	HSS $^{2310, 2330}$	118 *0.057*	324 *0.077*	-
Myrmeleotettix maculatus	Mottled Grasshopper	LC	HSS $^{2310, 2330}$	68 *0.033*	243 *0.058*	82 *0.060*
Oedipoda caerulescens	Blue Winged Grasshopper	LC	HSS $^{2310, 4030}$	112 *0.054*	296 *0.070*	189 *0.138*

3. Results

3.1. Point process analysis

Species occurrence sets showed spatial interaction at radii of 50 to 250 metres (Table B.1 and Fig. B.2 to B.11). Model covariates showed no multicollinearity (Table B.2; VIF < 3 and average Pearson correlations r = -0.007 ± 0.193, r = 0.023 ± 0.188 and r = -0.060 ± 0.261 in open, closed and anthropogenic landscape contexts respectively). Models fitted the data reasonably well, with no goodness-of-fit test (DCLF) rejecting the null hypothesis at a 0.01 significance level (Table B.1). Bird models performed better than most butterfly and grasshopper models, with the latter also presenting more variation in model performance (Table C.1).

3.2. Drivers of habitat suitability

For the general impact of heathland size, we found that larger heathlands were more suitable than intermediate or small heathland patches for all three taxonomic groups, especially in an anthropogenic landscape context (Fig. 2; Fig. 3). For the general impact of the landscape context, results in semi-natural (i.e. open or closed) contexts were mostly different from those in an anthropogenic context. Note that we could not assess the impact of the landscape context on habitat suitability parametrically due to correlations with all other predictors (Fig. B.1).

In the next paragraphs, we will demonstrate that the habitat suitability of the species in all three taxonomic groups was impacted by both vegetation structure and heathland heterogeneity, depending on heathland size and the landscape context. It is important to present these results in an integrated manner. Fig. 2 and Fig. 3 therefore show the impact of both vegetation structure and heathland heterogeneity on relative habitat suitability in the three landscape contexts and three classes of heathland size (section 2.3.1.b), yet in slightly different ways. Fig. 2 shows predictions of the relative habitat suitability across the environmental ranges of vegetation structure, heathland heterogeneity and heathland size at constant values of the sampling bias covariates (Warton et al., 2013). The figure allows to assess the combined impact of vegetation structure and heathland heterogeneity and is visually more intuitive, with colours representing a low or high relative habitat suitability (also see Fig. D.1 to D.3). Fig. 3 shows the estimates of the model coefficients (betas in Equation 5 in Appendix A.2) and allows to better interpret the relative importance of the impact of vegetation structure, heathland heterogeneity and heathland size on relative habitat suitability (also see Fig. D.4 to D.6). Predictions (Fig. 2) and estimates (Fig. 3) were averaged within the three classes of heathland size, i.e. small (≤ 10 hectares), intermediate (10-30 hectares) and large patches (> 30 hectares). We will discuss the aggregated results for all species and per taxonomic group in the main text and refer to Appendix D for the results per species.

The impact of vegetation structure and/or heathland heterogeneity on habitat suitability depended on the surrounding heathland size for most species in at least one landscape context (Appendices D and E). A high vegetation structure became more important at larger heathland sizes, but in small patches in a closed or open landscape context also a low vegetation structure could benefit habitat suitability, especially for birds and butterflies (Fig. 3). A high heathland heterogeneity mostly impacted habitat suitability positively (Fig. 2; Fig. 3) and became more important when heathland size increased in an open landscape context while being equally important across heathland sizes in a closed or anthropogenic context (Fig. 3).

Pooling all species (boxplots in Fig. 3), habitat suitability in an open landscape context was highest in large patches with high heathland heterogeneity or small patches with a low vegetation structure. In a closed landscape context, high heathland heterogeneity is beneficial, as are small heathland patches with a low vegetation structure and large patches with a high vegetation structure. In an anthropogenic landscape context, habitat suitability was highest in patches with a high heathland heterogeneity and a high vegetation structure, especially in large patches.

Bird habitat suitability was positively impacted by heathland heterogeneity, especially in large patches in a semi-natural landscape context. Habitat suitability further increased in small and intermediate patches/patch edges with a low vegetation structure in a semi-natural context, and in intermediate patches/patch edges and large patches with a high vegetation structure in an anthropogenic context. Butterfly habitat suitability was positively impacted by both a high vegetation structure and a high heathland heterogeneity in large patches in an open or anthropogenic context. In a closed context, a high vegetation structure increased habitat suitability in large patches and a high heathland heterogeneity did so in intermediate patches/patch edges. In small patches, habitat suitability for butterflies increased with a low vegetation structure, combined with low heathland heterogeneity in an open landscape context and high heathland heterogeneity in a closed context. Grasshopper habitat suitability was higher at a high vegetation structure, especially in intermediate patches/patch edges, and at a high heathland heterogeneity in a closed or anthropogenic landscape context.

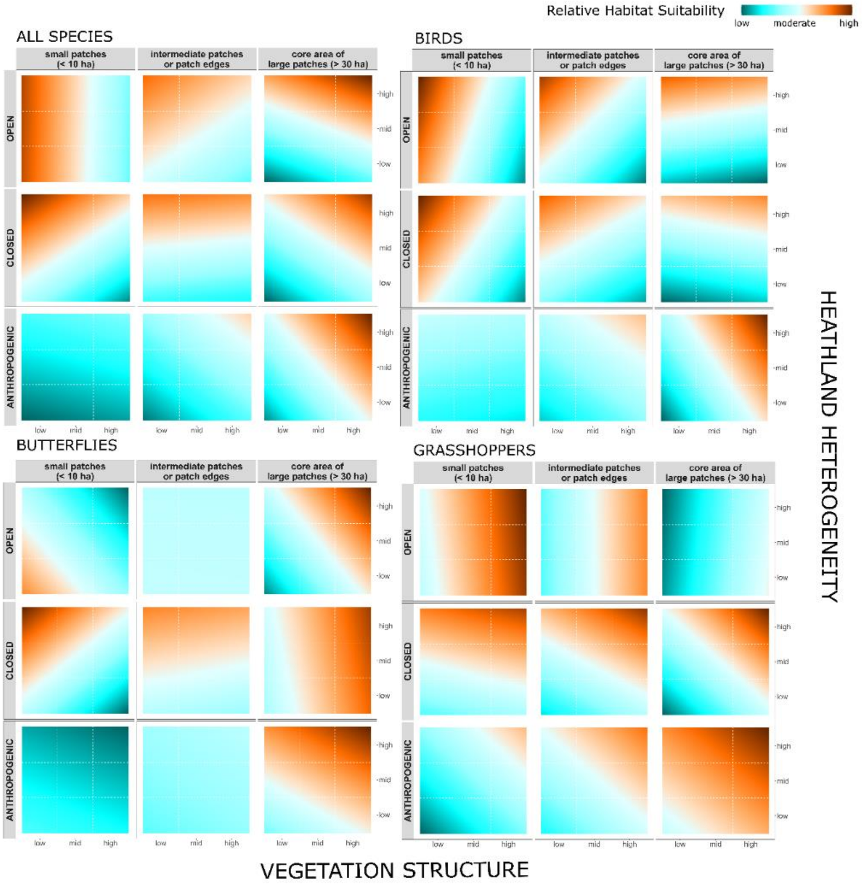

Fig. 2: Predictions of relative habitat suitability: - the impact of vegetation structure (x-axis) and heathland heterogeneity (y-axis) on relative habitat suitability in different landscape contexts, summarised in three classes of heathland size. Predicted intensities were first log-transformed to generate a linear output and then scaled and averaged across all considered dry heathland species and according to taxonomy in different landscape contexts (blue = low relative suitability, orange = high relative suitability). These values are the results of different Gibbs point process models with Geyer saturation process per landscape context, including two-way interactions between heathland size and vegetation structure/heathland heterogeneity. For four species (i.e. Caprimulgus europaeus, Callophrys rubi, Plebejus argus and Gryllus campestris), the model in the anthropogenic landscape context was omitted (see section 2.4).

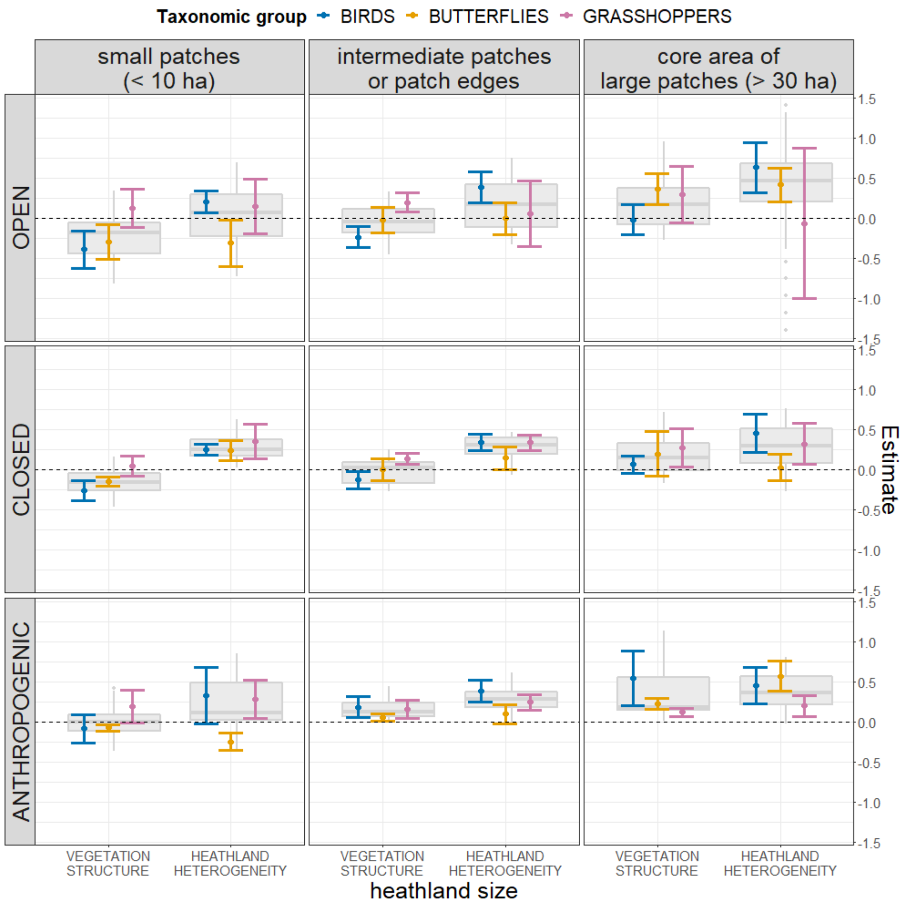

Fig. 3: Estimates of the model coefficients: the positive or negative impact of vegetation structure and heathland heterogeneity on relative habitat suitability (quantified by intensity) in different landscape contexts, summarised in three classes of heathland size. The distribution of the model coefficients is shown for all species (boxplots) and grouped according to taxonomy, with dots and error bars representing mean estimate values and standard deviations. These values are the results of different Gibbs point process models with a Geyer saturation process per landscape context, including two-way interactions between heathland size and vegetation structure/ heathland heterogeneity. For four species (i.e. Caprimulgus europaeus, Callophrys rubi, Plebejus argus and Gryllus campestris), the model in the anthropogenic landscape context was omitted (see section 2.4).

4. Discussion

By integrating opportunistic citizen science data and multispectral satellite data in point process models, we have substantiated the importance of vegetation structure for heathland fauna (Byriel et al., 2023; Maes et al., 2017b). At the same time, we highlighted some important considerations in human-dominated and fragmented landscapes (Maes et al., 2022), such as the impact of edge effects from the surrounding land use and the ecological needs of the considered taxonomic group. Quantifying vegetation structure and heathland heterogeneity in a standardized and spatially contiguous way through remote sensing enabled us to produce generalisable results, an important asset for biodiversity policy and conservation.

Habitat heterogeneity (larger scale) and vegetation structure (smaller scale) have been shown to increase niche availability and diversify environmental resources (e.g., MacArthur and Wilson, 1967). They usually positively affect habitat suitability and species diversity (Ampoorter et al., 2020; Cramer and Willig, 2002) although this relationship can also remain undetected or even be negative, depending on the spatial scale, the type of heterogeneity measure and the taxonomic group considered (Stein et al., 2014; Tews et al., 2004). Our results also demonstrated positive impacts of heathland heterogeneity and of vegetation structure in core areas of large heathland patches. In small and fragmented patches, however, vegetation structure was often negatively associated with habitat suitability for the studied birds and butterflies. This might be explained by the birds’ habitat preference and by the effect of micro-fragmentation, which implies that small-scale habitat heterogeneity can cause niche isolation for less mobile species (Laanisto et al., 2013). The four heathland birds prefer open to semi-open heathlands with occasional trees or shrubs as a viewing point for foraging or defending their territories. These relatively large species need large areas with a relatively low vegetation structure, especially in small and fragmented patches. While intuitively birds should be less affected by micro-fragmentation at the considered scale (0.25 hectares), lower food availability of species that are negatively affected by micro-fragmentation, such as invertebrates and plants (Laanisto et al., 2013; Tamme et al., 2010), could also explain the negative relationship with vegetation structure for birds.

Habitat edges induce edge effects that become stronger in smaller and more fragmented habitats (Ewers et al., 2007; Fahrig, 2003). This probably explains why we found the largest differences between landscape contexts in small patches and patch edges. Traditionally, habitat suitability was negatively associated with fragmentation (Hanski, 1998; MacArthur and Wilson, 1967), but the surrounding land cover might also increase habitat suitability for species that can benefit from edge effects (Dupont and Overgaard Nielsen, 2006; Evens et al., 2018; Maes et al., 2014; Pfeifer et al., 2017). Habitat edges in a semi-natural landscape context can provide resources for the inhabiting species such as shelter, nesting spots or foraging opportunities (Dupont and Overgaard Nielsen, 2006; Evens et al., 2018) and deliver specific habitat conditions such as forest ecotones (Maes et al., 2014; Moquet et al., 2018; Pfeifer et al., 2017). Small and isolated patches can thus have high habitat suitability (Wintle et al., 2019) if located in a resourceful landscape. The surrounding semi-natural land cover might even enhance the habitat heterogeneity to the point that maintaining characteristic heathland vegetation (i.e. dwarf shrubs, quantified by a low vegetation structure; Fig. 1c) will become relatively more important, especially for species that rely on them for food and reproduction (Byriel et al., 2023) such as Grayling (*Hipparchia semele*), European Stonechat (*Saxicola rubicola*) and Silver-studded Blue (*Plebejus argus*). In a closed landscape context that consists of forest only, however, maintaining heathland heterogeneity remains essential. This was also noted for large patches in an anthropogenic context, where maintaining a high vegetation structure and heathland heterogeneity appeared relatively more important compared to large patches in a semi-natural landscape context.

Butterflies are considered an umbrella taxon for insect conservation (e.g. van Swaay et al., 2006) and birds are often used as indicators of general habitat quality (Maes et al., 2009). Yet, results among taxonomic groups, even among invertebrates, showed dissimilarities (Fig. 2; Fig. 3). Taxonomic groups respond to different components of vegetation structure at different spatial scales (Atauri and De Lucio, 2001; Davies and Asner, 2014; de Vries et al., 2021; Tews et al., 2004). The impact of local vegetation structure on bird habitat suitability, for example, would not have been detected by large-scale measures of habitat heterogeneity and certainly not by those derived from coarse categorical land use maps (Coops and Wulder, 2019). Additionally, possible benefits of edges can be higher for taxonomic groups with larger (birds) or more mobile species (birds and butterflies) (Pfeifer et al., 2017) as opposed to taxonomic groups with small and less mobile species (grasshoppers). For grasshoppers, habitat suitability increased in intermediately large patches or patch edges with a high vegetation structure and at a high heathland heterogeneity in a closed or anthropogenic landscape context, as was observed in an earlier study for Blue Winged Grasshopper (*Oedipoda caerulescens*) and Mottled Grasshopper (*Myrmeleotettix maculatus*) (Schirmel et al., 2011). The three heathland grasshopper species are mostly found in relatively open habitats but need sufficient high grasses for food and shelter by bushes or small trees during very hot days (Willott, 1997). Additionally, the “enemy-free space hypothesis” states that prey species prefer dense vegetation with a high structure to escape from predators (Price et al., 1980). This was found to be true for large carabid beetles (Brose, 2003) and is probably also the case for grasshoppers in our study.

Although pooling species into taxonomic groups revealed some patterns regarding the impact of the environmental variables on habitat suitability, individual species might respond differently to multiscale vegetation management. Conservation planners must, therefore, consider additional knowledge on habitat requirements of dry heathland species, especially those of conservation interest. For example, bird habitat suitability was generally impacted positively by a high heathland heterogeneity, although this was less pronounced for European Nightjar (*Caprimulgus europaeus*) (Fig. D.4). This species requires complementary habitats for foraging (extensive grasslands) and breeding (heathlands) which may be separated by several kilometres (Evens et al., 2018). Habitat size and heterogeneity on a wider landscape scale will likely be more important than the heterogeneity of habitat subtypes within the heathland itself (Evens et al., 2021). Another example is the overall preference for a low vegetation structure in a closed landscape context for the Silver-studded Blue. This preference was also detected in intermediate and large patches, as opposed to the other two butterfly species which preferred a high vegetation structure in larger heathlands (Fig. D.5; Table E.1).

A low vegetation structure can indicate the presence of relatively young and low *Calluna vulgaris* and *Erica tetralix,* the preferred host plants of the Silver-studded Blue (Diemont et al., 2015).

Our results support that restoring and maintaining large and structurally complex habitats with patchy vegetation is a good approach for fauna conservation in heathlands (Byriel et al., 2023; De Blust, 2022; de Vries et al., 2021; Schirmel et al., 2011; van den Berg et al., 2001). The positive impact of an increased heathland size for most species is expected as habitat loss threatens biodiversity (Newbold et al., 2015). Furthermore, positive relationships between increased availability of a species' associated land use or habitat type and occurrence are common, especially for habitat specialists (Fahrig, 2003; Milanesi et al., 2017; Rutten et al., 2019; van den Berg et al., 2001). Heathland enlargement becomes especially important in an anthropogenic landscape context (i.e. urban and agriculture land use; Piessens et al., 2005; Worboys et al., 2010), due to negative edge effects and low suitability of the surrounding land use for species of conservation interest (Fletcher et al., 2018; Newbold et al., 2015; Olivier et al., 2016). Increasing heathland area can be achieved by restoring heathland habitat, for example by felling (non-native) coniferous forests. In regions with highly fragmented and isolated patches facing strong anthropogenic pressures, such as nitrogen deposition from agriculture or industry (Vogels et al., 2017), this can be challenging due to policy restrictions, budgetary limitations or land ownership (Diemont et al., 2015; Maes et al., 2022). In this light, it is essential to understand that even small patches can have adequate habitat suitability for typical (threatened) heathland species when habitat heterogeneity and/or vegetation structure are sufficiently high.

Increasing heterogeneity in nitrogen-polluted heathlands is often realised by large-scale removal of above-ground vegetation (e.g. by clearcutting, machine cutting or burning) or of both vegetation and soil top layers (i.e. sod-cutting or choppering) (De Blust, 2022). Those large-scale and intensive management practices homogenise the vegetation cover, lower vegetation structure and deplete nutrients from the soil, which is beneficial for restoring typical heathland vegetation (Jones et al., 2017; Schellenberg and Bergmeier, 2020), but can also have a detrimental effect on invertebrates and larger predators, such as birds, that feed on them (Maes et al., 2017b; Vogels et al., 2021, 2017). Therefore, intensified large-scale management practices should be avoided when possible, especially in and around (small) areas where species of conservation interest are known to be present.

The proxy that was used to quantify vegetation structure characterizes mainly heterogeneity in vegetation height with ancillary information on horizontal plant diversity (Farwell et al., 2021), yet both components are inextricably linked. Increasing the vertical complexity of vegetation cover at smaller scales will automatically allow for more plant diversity and can be achieved relatively fast, for example by removing above-ground vegetation and preventing grass encroachment of bare soil by mosaic mowing, cutting trees or low-intensity grazing, while allowing other patches to reach older successional stages (Byriel et al., 2023). While using a multivariate structural proxy has been shown to outperform single components of vegetation structure for estimating species distributions and diversity (e.g. Brose, 2003; Farwell et al., 2021), it also complicated the interpretation of which component of the proxy (heterogeneity in vegetation height or horizontal plant diversity) impacted habitat suitability. Combining a structural proxy with LiDAR-derived vegetation height, for example, might help to disentangle the individual impact of the components (Bergen et al., 2009; de Vries et al., 2021; Moudrý et al., 2022). Future research can also include microclimate data at fine scales obtained from remote sensing (Zellweger et al., 2019). This can, for example, shed further light on the importance of vegetation structure for invertebrates in heathlands as a regulator under climatic extremes (Maes et al., 2019b; Mantilla-Contreras et al., 2012; Schirmel et al., 2011; Schirmel and Fartmann, 2014).

We remain careful to generalise our definition of multiscale management to a ‘small versus large-scale approach’. We did find important indications that heathland size, the landscape context and taxonomy affect the scale at which heathlands are best managed. Additional findings from a sensitivity analysis (where vegetation structure and heathland heterogeneity are quantified at different spatial scales), however, could further support management recommendations and might highlight some keystone structures in heathland ecosystems (Tews et al., 2004). We also assumed that a species responded uniformly to the environmental gradients throughout the Campine region, as it is a region with similar biotic and abiotic conditions (Klijn and de Haes, 1994). We acknowledge, however, that on larger scales, the impact of climatic variables on habitat suitability becomes more prominent or species populations might respond differently to similar local environmental conditions (Chen et al., 2020). For example, heathland butterflies responded similarly to environmental conditions within the Campine region (Vanreusel et al., 2007) but might respond differently to spatial structure in other regions (De Ro et al., 2021; Schirmel and Fartmann, 2014). Although the presented methods are applicable in any region where both species occurrence and remote sensing data are available at high resolutions, they might not be transferable to every habitat type, as quantifying habitat heterogeneity and vegetation structure possibly needs different approaches or considerations. For example, habitat heterogeneity in farmland is not only impacted by variability in habitat subtypes (such as arable land, cultural grassland and orchards) but also by crop configuration and composition (Fahrig et al., 2011) and by the presence of small landscape features such as hedgerows or flower strips (Dochy, 2014).

5. Conclusion

Integrating fine-scaled opportunistic citizen science data and remote sensing data with point process models is a promising advance in biodiversity conservation monitoring. By including heathlands of different sizes and the landscape context, management recommendations for fauna conservation could be formulated in highly fragmented landscapes. In summary, we recommend restoring and maintaining large and structurally complex heathlands with patchy vegetation. Conservation should also include action plans to connect fragmented heathlands (e.g. by cutting down pine plantations). When sufficient natural resources are available in the direct (semi-natural) environment of smaller heathlands (e.g. for foraging or nesting), heathland management plans should also emphasize the importance of maintaining characteristic dry-heathland shrub vegetation. In anthropogenic landscapes, on the other hand, simply increasing the structural complexity of patches without increasing their surface area might not be enough to avoid the local extinction of species of conservation interest. This is probably the largest challenge for conservation management, as enlarging heathlands in human-dominated landscapes will need the field experience of local managers, the sensitisation of citizens and additional actions and funding at the policy level. We further emphasize the importance of using multiple species from different taxa as a conservation umbrella, which has become more and more feasible considering the unprecedented quantity of species occurrence data collected on citizen science data platforms.

DECLARATIONS

Ethical approval (not applicable)

Competing interests (not applicable)

Authors' contributions

Camille Van Eupen: conceptualization, methodology, software, validation, formal analysis, writing – original draft, visualization; Dirk Maes: conceptualization, writing – review & editing, supervision; Stien Heremans: conceptualization, writing – review & editing; Kristijn Swinnen: data curation, writing – review & editing; Ben Somers: conceptualization, writing – review & editing, supervision; Stijn Luca: conceptualization, writing – review & editing, supervision. All authors contributed critically to the drafts and gave final approval for publication.

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Availability of data and materials

The cleansed datasets of the ten selected species will be made available in Dryad Digital Repository. All species data are available on request at Natuurpunt Studie (natuurdata@natuurpunt.be).

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