

Alma Mater Studiorum Università di Bologna
Archivio istituzionale della ricerca

Combining climate, land use change and dispersal to predict the distribution of endangered species with limited vagility

This is the final peer-reviewed author's accepted manuscript (postprint) of the following publication:

Published Version:

Della Rocca, F., Milanese, P. (2020). Combining climate, land use change and dispersal to predict the distribution of endangered species with limited vagility. JOURNAL OF BIOGEOGRAPHY, 47(7), 1427-1438 [10.1111/jbi.13804].

Availability:

This version is available at: <https://hdl.handle.net/11585/927452> since: 2024-05-15

Published:

DOI: <http://doi.org/10.1111/jbi.13804>

Terms of use:

Some rights reserved. The terms and conditions for the reuse of this version of the manuscript are specified in the publishing policy. For all terms of use and more information see the publisher's website.

This item was downloaded from IRIS Università di Bologna (<https://cris.unibo.it/>).
When citing, please refer to the published version.

(Article begins on next page)

Combining climate, land use change and dispersal to predict the distribution of endangered species with limited vagility

Running title: **Integrating climate, land use change and dispersal**

DELLA ROCCA FRANCESCA¹, MILANESI PIETRO²

¹ *Dipartimento di Scienze della Terra e dell'Ambiente, Università degli Studi di Pavia, Via Ferrata 1, 27100 Pavia, Italy. fdellarocca@gmail.com*

² *Swiss Ornithological Institute, Seerose 1, 6204 Sempach, Switzerland.*

Corresponding author: Francesca Della Rocca fdellarocca@gmail.com

ACKNOWLEDGEMENTS

We thank Prof. Francesco Bracco, manager of the Riserva Naturale Integrale Bosco Siro Negri, who supported part of this research through funds from the Italian Ministry of the Environment and Protection of Land and Sea. We thank the European Environment Information and Observation Network (EIONET) for making their data publicly available; Peter Verburg, who kindly provided the land use change scenario maps for the year 2040, and Stephanie Witczak for revising and improving this manuscript. The comments of the Editor-in-Chief Jon Sadler, the Associate Editor Jenny McGuire, as well as those of Jakub Horák and an anonymous reviewer greatly helped improving this paper.

ABSTRACT

Aim. To investigate the effects of both climate and land use change on the distribution of endangered beetle species with limited vagility, accounting for and ignoring species-specific dispersal when projecting species distribution.

Location. Europe.

Time period. 2007-2012; 2040-2050.

Taxon. Saproxylic beetles.

Methods. Using current (2007-2012) species occurrences, we predicted the distribution of six endangered saproxylic beetles under a range of future (2040-50) climate and land use change scenarios in Europe by using ecological niche models.

Results. Our results showed that, while all the species considered would increase their distribution, areas both accessible and suitable would range between only 5% and 38% of the total potential available area, depending on the species and the future scenarios considered.

Main conclusion. We strongly encourage researchers and conservationists to combine climate and land use change with dispersal when projecting species distribution under future scenarios. This should avoid misleading predictions and overestimation, as climate and land use change generate differing effects depending on the inclusion or exclusion of species dispersal abilities.

KEYWORDS: ecological niche models, future scenarios, global change, macroecology, *n*-dimensional hypervolume, saproxylic beetles, species traits.

1. INTRODUCTION

Many rare species are dispersal-limited, exhibiting a low probability of colonizing new sites and establishing viable populations (Baur, 2014). The current and future distributions of these species are highly influenced by human-related environmental change, such as climate and

land use change, creating insurmountable barriers to their dispersal across space and time (Jaeschke et al., 2013). Although both climate and land use change are considered dominant drivers of biodiversity at a global scale (Barbet-Massin et al., 2012; Pimm et al., 2014), only the former has been widely used as a predictor for modelling future species distributions (e.g. Markovic et al., 2014; Titeux et al., 2016). Indeed, during the last 25 years, more than 85% of published research in this field explored the impact of future climate change alone on biodiversity (Titeux et al., 2016). Moreover, in the few studies that considered the impact of land use change on species distribution (Radinger et al., 2016; Milanesi et al., 2017a), the combined effect with climate change was rarely investigated (Mantyka-Pringle et al., 2014; Radinger et al., 2016). Since species range shift varies depending on site-specific climate-land use combinations, neglecting the effect of land use change could lead to biased predictions of species distributions under future conditions. Especially, this holds true for species with limited dispersal capacities, for which even minimal land use changes can dramatically impact their colonization capacity (Velo-Antón et al., 2013; McCauley et al., 2014).

Actually, the Biotic-Abiotic-Mobility framework ('BAM'; Soberón & Peterson, 2005) highlighted also the importance of species-specific dispersal characteristics in determining species distribution (De Marco et al., 2011; Nobis & Normand, 2014; Vasudev et al., 2015). Incorporating dispersal in ecological niche models (ENMs) improves predictions for current and future species occurrence compared to standard ENMs (Engler & Guisan, 2009; Smolik et al., 2010; Vasudev et al., 2015). However, most studies developing ENMs to predict the distribution of species under future scenarios have not incorporated species-specific dispersal abilities, relying on overly simplistic conceptualizations of dispersal (Franklin, 2010; Peterson et al., 2011).

Taking into account these limitations, in this study we (i) investigated the effect(s) of climate and land use change on species with short-distance dispersal abilities and (ii) defined areas that should be accessible to them in the future. We used the current (2007-2012)

occurrences of six endangered saproxylic beetles, widely considered to be umbrella species for biodiversity conservation (Ranius, 2002; Buse et al., 2007; Russo et al., 2011; Campanaro et al., 2011; Solano et al., 2013; Bełcik et al., 2019), to develop ENMs using current climate and land use conditions. We then projected the distributions of our target species under future scenarios to estimate species potential occurrences in the years 2040-50 (based on four different climate and land use change scenarios). Finally, taking into account the species-specific dispersal abilities of our target species, we limited their distributions in 2040-50 to areas accessible to them across Europe.

2. METHODS

2.1 Study area and species data

Our study area consisted of all continental European countries (excluding Belarus, Moldova, Russia and Ukraine, because of the lack of data in national biodiversity repository servers and in other official archives; Fig. 1). The study area ranges from 0 to 4,810 m a.s.l., and is characterized by forests (33.3% of the total area), croplands (32.4%), shrub-lands (11.5%) and, to a lesser extent, grasslands (7.8%) and human settlements (4%).

We selected six threatened species of saproxylic beetles (out of the 21 saproxylic beetles species listed in the EU habitat directive), namely *Cerambyx cerdo*, *Cucujus cinnaberinus*, *Lucanus cervus*, *Morimus funereus*, *Osmoderma eremita* and *Rosalia alpina*, for which information on observed dispersal distances were available in literature (Table S1). We obtained species occurrence data for the period 2007-2012 from (i) the European Environment Information and Observation NETwork (EIONET, 2013) Central Data Repository server (<http://cdr.eionet.europa.eu/>), (ii) the Swiss Biological Records Center (<http://lepus.unine.ch/carto/>) and (iii) the Norwegian Taxonomy Initiative (<http://www.biodiversity.no/>). All species occurrences were resampled at the same spatial resolution of EIONET data (10 × 10 km grid cell size; Fig. 1), resulting in a total of 4,310

cells for *C. cerdo*, 3,561 for *C. cinnaberinus*, 11,535 for *L. cervus*, 2,211 for *M. funereus*, 5,248 for *O. eremita* and 2,628 for *R. alpina*. We tested for overestimation of saproxylic species occurrences calculating a modified version of the Multivariate Environmental Similarity Surface (mMESS, Milanesi et al., 2017b). We find consistent results (Fig. S1-6) and thus, we included all data collected in the further analysis.

2.2 Species ecological requirement

The species considered in this study (Table S1) are highly susceptible to both climate and land use changes (Gough et al., 2015). In fact, being highly specialized wood-living beetles, their survival is, above all, linked to the availability of suitable forest habitat. As such, their ability to respond to climate change is mainly influenced by habitat degradation (Filz et al., 2013; Ball-Damerow et al., 2014). Moreover, the study species are characterized by low dispersal distances (Schiegg, 2001), and inhabit most of the European deciduous old-growth forests currently threatened by large-scale human disturbance (Wirth et al., 2009; Stokland et al., 2012).

C. cerdo is widespread in most parts of Europe, but more common in the Mediterranean regions, generally associated with oak forests consisting of mature or partially dead, and sun-exposed trees (Sama, 1988). This species occurs in semi-open forest patches of lowland and hilly forests (Redolfi de Zan et al., 2017).

C. cinnaberinus is a poorly known species (Horák, 2011) limited to Europe, scattered distributed throughout the continent, probably because of past population decline and local extinctions (Horák & Chobot, 2009; Horák et al., 2010). This species live under the bark of dead tree trunks (Horák et al., 2008) in lowland poplars and willows forests or partially-mixed mountain forests (e.g.; Horák et al., 2010).

L. cervus is widely spread across Europe and is associated with mature deciduous forests. It can be found especially in lowland and medium-altitude oak woodlands, where the

saproxylic larvae feed on rotten deadwood at ground level (Campanaro et al., 2011). This species may also be common in urban habitats (e.g. city parks, private gardens; Hawes, 2008; Harvey et al., 2011).

M. funereus inhabits mature beech, poplar and oak forests (Sama, 2009) within a relatively narrow geographical zone in south-eastern Europe (Jurc et al., 2008; Carpaneto et al., 2015). Even though their preferred habitats are mature forests, populations of this species often occur in coppiced stands, characterised by old stumps and decaying wood on the ground (Hardersen et al., 2017).

O. eremita is a genus that includes many species. Based on genetic studies (Audisio et al. 2007, 2009), there is now consensus that *Osmoderma eremita* is just one of four species occurring in Europe. However, in our analysis, we considered all these species as belonging to *Osmoderma eremita*. It is generally associated with hollow veteran trees (Ranius & Hedin, 2001; Svensson et al., 2011), of the ecotonal areas and clearings of mature forests, agricultural and urban landscapes (Maurizi et al., 2017).

R. alpina lives mainly in the mountainous regions of central and southern Europe and is associated with beech forests, but also maples and elms (Bosso et al., 2013; Lachat et al., 2013). It prefers open and semi-open woodlands (Russo et al., 2011), reproducing mainly in mature, dead (or declining), and sun-exposed trees (Campanaro et al., 2017a).

2.3 Predictor variables

For the period 2007-2012, we considered a total of 28 predictors of species occurrence, accounting for the assumed habitat characteristics of the target species, and for which continuous spatial data were available for the entire study area (Table S2). We considered two topographic variables (ASTER GDEM; gdem.ersdac.jspacesystems.or.jp), seven land use variables (CORINE Land Cover 2012; <https://land.copernicus.eu/pan-european/corine-land-cover/clc-2012>) and the Euclidean distances to human settlements (Table S2). Moreover, we

considered 19 bioclimatic variables (<http://www.worldclim.org>; Table S2). All predictors were resampled at a 10×10 km grid cell size.

For the period 2040-50, we considered four different scenarios of climate and land use change. Specifically, thanks to recent advances in the development of socioeconomic storylines and their potential effect on future land use patterns, we used four land use change scenarios addressing both changes in land cover and land use intensity, reflecting socioeconomic, cultural, political, and technological changes in the EU (Stürck et al., 2015; data provided by the authors). These scenarios, namely Libertarian Europe (A1), Eurosceptic Europe (A2), Social Democracy Europe (B1) and European Localism (B2), represent strong, high, moderate and low economic interventions and growth, respectively (see Table S3 and Stürck et al., 2015 for details on the scenarios).

Similar to Ihlow et al. (2016), we considered four climate change scenarios (Representative Concentration Pathways, RCPs) for the year 2050 averaging 11 general circulation models (GCMs: BCC-CSM1-1, CCSM4, GISS-E2-R, HadGEM2-AO, hadGEM2-ES, IPSL-CM5A-LR, MIROC-ESM-CHEM, MIROC-ESM, MIROC5, MRI-CGCM3 and Nor-ESM1-M). These scenarios were obtained from the fifth assessment of the Intergovernmental Panel for Climate Change (IPCC AR5WG1 2014; <http://www.ipcc.ch>). The selected RCPs represent four possible greenhouse gas emission trajectories, including low (RCP 2.6), moderate (RCP 4.5), high (RCP 6) and strong (RCP 8.5) increases in global radiative forcing (Ihlow et al., 2016). Based on these scenarios, we used the same predictors for the period from 2007-2012, available at a 10×10 km grid cell size (the same resolution as current climatic conditions) from the Worldclim website (<http://www.worldclim.org>).

2.4 Data analysis

To avoid multi-collinearity among predictors, we estimated the variance inflation factor (VIF) for each predictor. Thus, in the further analysis we retained 13 (ecologically relevant;

Thomaes et al., 2008; Bosso et al., 2013; Campanaro et al., 2017b) out of 28 predictors due to VIF values lower than three (i.e. poorly correlated with other predictors; Zuur et al., 2010; Table S2).

Thus, we applied a recently developed ENMs, namely "*n*-dimensional hypervolume" (R package HYPERVOLUME v. 2.0.11; Blonder et al., 2014; 2017a), considering current species occurrences, climate and land use conditions but projecting on different future (2040-50) climate and land use change scenarios (see Table S4 for details on model parameters). *N*-dimensional hypervolume has several benefits compared to other extant ENMs as it (i) considers all the dimensions (predictors) to measure the volume of a high-dimensional shape (including holes or other complex geometrical features; Blonder et al., 2014; 2017a), (ii) allows delineation of the entire hypervolume without generating unbounded volumes (Blonder et al., 2014; Peterson et al., 2011) and (iii) while relying on presence-only data, correctly classified false absences through robust bandwidth estimation methods (e.g. Silverman estimator; Blonder et al., 2017b).

Similar to Jaeschke et al. (2013), we identified suitable areas accessible to our target species in the future multiplying the yearly dispersal distance of each species (Table S1) for the total number of years ($n=33$, from 2013 to 2045, average of 2040-50). The resulting distances were divided by the number of years larvae would take to develop in the adult stage (Table S1) plus one and then used to define accessible areas around current occurrences.

3. RESULTS

Without taking into account species-specific dispersal abilities, we found that suitable areas for the saproxylic beetles for the period 2040-50 would vary between 3,345,200 km² and 4,661,300 km², corresponding to the scenario A2 – RCP 2.6 for *M. funereus* and scenario B2 – RCP 6 for *L. cervus*, respectively (Table 1). Specifically, suitable conditions for *C. cerdo* would occur all over Europe, mainly in the central-eastern part of the continent, while those

for *C. cinnaberinus* would occur in many areas of central-eastern Europe (excluding Switzerland). For *L. cervus* suitable areas would be mainly located from eastern to western Europe (excluding the southern part of the European continent), while those of *O. eremita* would be in central-eastern Europe, including Sweden and Finland. Finally, suitable areas for *M. funereus* would be widely located across central-eastern Europe, while *R. alpina* would be distributed throughout Europe, though most concentrated in France, Lithuania and Latvia.

When examining the suitable areas available for the six modelled species (generated without accounting for dispersal abilities) we found a higher impact of climate change relative to land use change on the distribution of *C. cerdo*, *C. cinnaberinus*, *M. funereus* and *R. alpina*. For these species we found a direct relationship between climate change increase and the availability of suitable areas (Fig. 2). Conversely, the suitable area of *L. cervus* and *O. eremita* is more susceptible to land use change than climate change, reaching the maximum extent in land use change scenarios B2 and A1, respectively (Fig. 2)

When we combined climate and land use change with species-specific dispersal abilities, we found that from 2040-50 accessible suitable areas for the six saproxylic beetles would range between only 5.05% and 38.98% of the total available suitable areas calculated previously (Table 2; Table S5). When considering dispersal abilities, *C. cerdo* would reach 24% of its suitable area available from 2040-50 (Fig. S7), while *C. cinnaberinus* would reach only 13% (Fig. S8), mainly located in Eastern Europe, Fennoscandian and Baltic countries. *L. cervus* would reach the largest percentage of its suitable area available from 2040-50 in relation to the other species considered, corresponding to 36 – 38% (Fig. S9), while *M. funereus* would reach only 5% of its suitable area for this timeframe (Fig. S10), mainly located in Slovenia, Hungary, Bulgaria, Southern Romania and Northern Greece. Finally, *O. eremita* would reach 26% of its suitable areas available in 2040-50 (Fig. S11), mainly located in central Europe, while *R. alpina* would reach 14% of its suitable area available during this

timeframe (Fig. S12), mainly located in the mountainous and hilly areas of Southern Europe (excluding Bosnia, Montenegro, Albania and Kosovo).

When including species-specific dispersal abilities we generally found a higher impact of land use change on the distribution of all six saproxylic beetles compared to when dispersal abilities were excluded (Fig. 3). Specifically, the highest percentage of accessible suitable areas for *C. cerdo* and *M. funereus* would be reached in the context of land use change scenario B1 while for *C. cinnaberinus* and *R. alpina*, it would be land use change scenario B2 (Fig. 3). Finally, for *L. cervus* and *O. eremita*, the highest percentage of available suitable area reached corresponds to the A1 land use change scenario (Fig 3). Generally, an increase in greenhouse gas emissions corresponds to an increase in the available suitable areas for *C. cerdo*, *C. cinnaberinus*, *L. cervus* and *R. alpina*, while in a decrease *O. eremita* (Fig 3). However, climate change would have a weak effect on the available suitable areas reached for *M. funereus* (Fig. 3).

4. DISCUSSION

Our results highlighted differences in the predicted distributions of saproxylic beetles given the differing effects that climate and land use can generate when accounting for or ignoring species dispersal abilities. Land use change was found to be the main constraint to species distributions when accounting for dispersal, while climate change when ignoring dispersal.

4.1 Distribution of saproxylic species considering future climate and land use changes

Our modelled species' ranges will grow in the future due to the combined effects of changing temperatures and precipitations (due to greenhouse gas emissions increase) and an expansion of forested areas (occurring in all the land use scenarios considered). However, while climate change has a similar effect on most of the investigated species, the effect of land use change is more species-specific. This is mainly due to the different landscape variables considered: the

expansion of grasslands, the abandonment of grasslands resulting in forest expansion, the expansion of human settlements, the intensification of forest management, and the resulting habitat diversity and the intensity of their exploitation (Stürck et al., 2015).

4.2 Accessible vs. available suitable areas under future climate and land use change scenarios

We found that none of the saproxylic beetles considered in this study will reach all of the suitable areas available in 2040-50, due to the limited dispersal abilities characterizing these species. Flightless species with narrow geographic distributions would be able to occupy sometimes as little as 5% of the suitable area available to them in 2040-50, as is the case with *M. funereus*. Conversely, widespread species might be able to cover as much as 38% of the suitable area available to them in 2040-50, as is the case with *L. cervus*. Thus, species-specific dispersal abilities greatly limit the ability to colonize new suitable areas.

Without accounting for species-specific dispersal abilities, we generally found a strong effect of climate change when predicting future species distributions (except for *L. cervus* and *O. eremita*). Indeed, for most of our species, suitable areas would increase with increasing greenhouse gas emissions, indicating better climatic conditions for their occurrence in the future, supporting a general expansion of many species of saproxylic beetles to higher elevations and latitudes in Europe. However, suitable areas of *L. cervus* and *O. eremita* would be mainly affected by land use change: the future scenarios “European localism” (B2 – low economic interventions and growth) and “Libertarian” (A1 – strong economic interventions and growth) predicted the maximum distribution expansion of these species, respectively.

When accounting for species-specific dispersal abilities, we found a weak effect of climate change and a rather strong effect of land use change. Indeed, we found a clear, inverse relationship between the amount of suitable accessible areas and the degree of land use change for *C. cinnaberinus* and *R. alpina*, for which the scenario “European localism” (B2 – low economic interventions and growth) predicted the highest expansion of suitable

accessible areas. This land use change scenario, followed by “Social Democracy Europe” (B1 – moderate economic interventions and growth), predicted the maximum expansion for *C. cerdo* and *M. funereus*. The main differences among these two scenarios concern the extent of grasslands, croplands (not included in the ENMs due to multi-collinearity, but see below), forests and the intensity of their exploitation (Table S3; Stürck et al., 2015). However, while the intensification of forest management, predicted for both the B1 and B2 scenarios, would present a threat for saproxylic beetles (Jonsson et al., 2005; 2006), the increase in forest cover (more than 170,000 km² of current croplands would be abandoned and return naturally to forests or grasslands; Stürck et al., 2015) would increase the extent of suitable habitat for our target species. Moreover, the de-intensification of grasslands predicted for both the B1 (in combination with human settlement contraction) and B2 scenarios (i.e. reduction of livestock density to sustainable grazing) represent a positive change for saproxylic beetles (Russo et al., 2011).

Although, at a broad scale, climate change is expected to be the main constraint for many species distributions, here we found that this is only true for *C. cinnaberinus*, *R. alpina* and *C. cerdo* in the unrealistic scenario of unlimited dispersal. Conversely, we found that land use change affects all our modelled species. In particular, *L. cervus* and *O. eremita* distributions are affected by land use change in both the cases of limited and unlimited dispersal, while those of *C. cerdo*, *C. cinnaberinus*, *M. funereus* and *R. alpina* are affected by land use change only upon taking their dispersal abilities into account. The strong effect of land use change on these saproxylic beetles could be explained by the high dependence of these species on the amount, availability and distribution of deadwood, which, in turn, is closely linked to forest use (Bradford et al., 2014; Mazziotta et al., 2016).

4.3 Caveats

307 We followed the approach of Jaeschke et al. (2013) for estimating yearly dispersal distance of
308 saproxylic beetles. While we estimated yearly dispersal distance by dividing the maximum
309 dispersal distance of each species by the number of years larvae would take to develop in the
310 adult stage plus one, this likely resulted in overestimation of the distance these species are
311 able to cover. Because of the lack of information on species-traits for our target species, we
312 assumed that the probability of colonization was 1 rather than 0. In doing so, we estimated the
313 maximum dispersal probability without including any species-traits in the model. Measuring
314 the real dispersive capacity of a species is very complex (Trakhtenbrot et al., 2005). In a
315 variety of actively dispersing invertebrates, precise estimations of long-distance dispersal
316 ability are based on morphological characteristics, such as body size or wingspan (Merckx &
317 VanDyck, 2002; Cizek, et al., 2006), or on life-history traits (Sutherland et al., 2000).
318 However, in many cases, such qualitative proxies do not prove informative about dispersal
319 ability, due to the uncertainty involved in the identification of the dispersal distance
320 mechanism (Trakhtenbrot et al., 2005). As such, often quantitative models are preferred
321 (Trakhtenbrot et al., 2005). Some used data on variables affecting dispersal to predict the
322 magnitude and frequency of long-distance dispersal, including changes in parameter values
323 caused by human impacts threatening biodiversity (Trakhtenbrot et al., 2005). However,
324 because of the high variability in behavioural and environmental factors (Higgins et al.,
325 2003), these models tend to be case-specific (e.g. South & Kenward, 2001; Macdonald &
326 Rushton, 2003) and often are applied on passive dispersers (Nathan et al., 2002) rather than
327 active animals, such as our target species. For many active dispersers, including insects,
328 phenomenological models are often used (Trakhtenbrot et al., 2005). These approaches can be
329 applied if previous population demographic studies are available, for example, capture-mark-
330 recapture studies that can provide presence/absence or dispersal distance data for the species
331 studied (Kuras et al., 2003). The quality and quantity of the necessary data needs to be
332 thoroughly assessed to make accurate predictions for specific species in a landscape context

(Whitmee & Orme, 2012). For most studies on species distribution these data are not available (Nathan, 2002).

Despite its limitations, we felt that the approach of Jaeschke et al. (2013) can be reasonably applied to our target species for two main reasons. First, the dispersal distances reported in this study are most likely underestimated. Most field studies on saproxylic dispersal distance focus on one or few populations (Chiari et al., 2013; Torres-Vila et al., 2017), and are spatially limited (Rossi De Gasperis et al., 2016; Drag et al., 2011) leading to an underestimation of the real distances an individual might be able to cover. This underestimation could be partially compensated by overestimation in our model. Second, although we may overestimate dispersal in our target species, our resulting spatial predictions are still very useful. In fact, they are more accurate than the unconstrained or no-dispersal scenarios often incorrectly assumed in most of the ENMs predicting the impact of climate change on species distributions (Araújo & Rahbek, 2006; Broennimann et al., 2006; Botkin et al., 2007).

While deadwood is the main resource affecting saproxylic beetle occurrence (Stokland & Siitonen, 2012; Milberg et al., 2016) and can influence microclimatic parameters (e.g. hollow trees, Ranius, 2002; Pilskog et al., 2016), we couldn't include deadwood-related variables in our analyses as deadwood amount and forest management intensity data is lacking at a continental scale (as well as a global scale). Actually, deadwood availability depends on forest management, which alter its distribution and abundance in a quite unpredictable way (i.e. much more than climate and land use; Della Rocca et al., 2018). Thus, simulated future scenarios of deadwood amount/forest management intensity would be random and not based on accurate data and validations. Hence, assuming that climate and land use change are the main driving forces available to model species distribution (Walther et al., 2002; Hitch & Leberg 2007; Guo et al., 2018), we can identify those areas where saproxylic species would exist should the deadwood required for their survival be present.

Finally, while the monitoring our target species is mandatory for EU Member States and regularly carried out in Switzerland and Norway, there are currently no standardized monitoring protocols at a continental scale (Campanaro et al., 2016). However, the methods used to sample our target species (e.g. attractive pheromones, baited traps, direct observations along transects) are the same within our study area (Campanaro et al., 2016; Maurizi et al., 2017) and thus comparable among different countries. Indeed, conservation actions at the continental scale within the EU are based on these data, collected from 28 countries.

5. CONCLUSIONS

We developed this study with the observation that most of the published papers aiming to predict species distribution under global change often lack the inclusion of species-specific dispersal abilities. Surprisingly, this holds true also when dealing with species with low vagility for which the dispersal is obviously the main factor limiting their distribution (McCauley et al., 2014). Thus, in this study we showed the importance of dispersal ability in species distribution modeling and how the effect of climate and land use change differed if dispersal abilities are taken into account or not. Moreover, we showed that combining climate and land use change scenarios with species-specific dispersal distances, resulted in more accurate and realistic projections of species distributions. Our findings are very important especially because poor dispersers suffer of high risk of extinction (Beissinger, 2000) due to their limited ability to move away from unsuitable habitat or climatic conditions. Saproxylic beetles are emblematic from this point of view. Through our approach we were able to accurately identify areas with fundamental species-specific resources, such as ancient woods, forest remnants, hedgerows and old deciduous tree and to provide tools for establishing protected areas and/or extending already existing ones. Concluding, we encourage researchers and conservationists to follow our approach when species dispersal

information is available to avoid misleading and overestimated predictions, providing support for conservation actions on these species and their habitats.

DATA AVAILABILITY STATEMENT

Species occurrence data used in this study is freely available at the following links:

<http://cdr.eionet.europa.eu/>, <http://lepus.unine.ch/carto/> and <http://www.biodiversity.no/>.

GIS layers related to bioclimatic predictors for both current and future scenarios used in this study are available at <http://www.worldclim.org>. Current land use GIS layers are available at <https://land.copernicus.eu/pan-european/corine-land-cover/clc-2012>, while future land use scenarios are freely available at <http://labs.kh.hercules-landscapes.eu/labs/themeLD.html>.

REFERENCES

- Araújo, M.B. & Rahbek, C. (2006). How does climate change affect biodiversity? *Science*, **313**, 1396–1397.
- Audisio, P., Brustel, H., Carpaneto, G.M., Coletti, G., Mancini, E., Piattella, E., Trizzino, M., Dutto, M., Antonini, G. & De Biase, A. (2007) Updating the taxonomy and distribution of the European *Osmoderma*, and strategies for their conservation (Coleoptera, Scarabaeidae, Cetoniinae). *Fragmenta Entomologica*, **39**, 273–290.
- Audisio, P., Brustel, H., Carpaneto, G. M., Coletti, G., Mancini, E., Trizzino, M., Antonini, G. & De Biase, A. (2009). Data on molecular taxonomy and genetic diversification of the European Hermit beetles, a species complex of endangered insects (Coleoptera: Scarabaeidae, Cetoniinae, *Osmoderma*). *Journal of Zoological Systematics and Evolutionary Research*, **47**, 88-95.
- Ball-Damerow, J.E., M’Gonigle, L.K. & Resh, V.H. (2014). Changes in occurrence, richness, and biological traits of dragonflies and damselflies (*Odonata*) in California and Nevada over the past century. *Biodiversity and Conservation*, **23**, 2107–2126.

410 Barbet-Massin, M., Thuiller, W. & Jiguet, F. (2012). The fate of European breeding birds
 411 under climate, land-use and dispersal scenarios. *Global change biology*, **18**, 881–890.

412 Baur, B. (2014). Dispersal-limited species—a challenge for ecological restoration. *Basic and*
 413 *Applied Ecology*, **15**, 559–564.

414 Beissinger, S.R. (2000). Ecological mechanisms of extinction. *Proceedings of National*
 415 *Academy of Science USA*, **97**, 11688–11689.

416 Bełcik, M., Goczał, J., & Ciach, M. (2019). Large-scale habitat model reveals a key role of
 417 large trees and protected areas in the metapopulation survival of the saproxylic specialist
 418 *Cucujus cinnaberinus*. *Biodiversity and Conservation*, 1-21.

419 Blonder, B., Lamanna, C., Violle, C. & Enquist, B.J. (2014). The n-dimensional
 420 hypervolume. *Global Ecology and Biogeography*, **23**, 595–609.

421 Blonder, B., Morrow, C.B., Maitner, B., Harris, D.J., Lamanna, C., Violle, C., Enquist, B.J. &
 422 Kerkhoff, A.J. (2017a). New approaches for delineating n-dimensional hypervolumes.
 423 *Methods in Ecology and Evolution*, **9**, 305–319.

424 Blonder, B., Lamanna, C., Violle, C. & Enquist, B.J. (2017b). Using *n*-dimensional
 425 hypervolumes for species distribution modelling: A response to Qiao et al. (). *Global*
 426 *Ecology and Biogeography*, **26**, 1071–1075.

427 Bosso, L., Rebelo, H., Garonna, A.P. & Russo, D. (2013). Modelling geographic distribution
 428 and detecting conservation gaps in Italy for the threatened beetle *Rosalia alpina*. *Journal*
 429 *for Nature Conservation*, **21**, 72–80.

430 Botkin, D.B., Saxe, H., Araujo, M.B., Betts, R., Bradshaw, R.H.W., Cedhagen, T., Chesson,
 431 P., Dawson, T.P., Etterson, J.R., Faith, D.P., Ferrier, S., Guisan, A., Hansen, A.S., Hilbert,
 432 D.W., Loehle, C., Margules, C., New, M., Sobel, M.J. & Stockwell, D.R.B. (2007).
 433 Forecasting the effects of global warming on biodiversity. *Bioscience*, **57**, 227–236.

434 Bradford, M.A., Warren II, R.J., Baldrian, P., Crowther, T.W., Maynard, D.S., Oldfield, E.E.,
 435 Wieder, W.R., Wood, S.A. & King, J.R. (2014). Climate fails to predict wood
 436 decomposition at regional scales. *Nature Climate Change*, **4**, 625.

437 Broennimann, O., Thuiller, W., Hughes, G., Midgley, G.F., Alke-made, J.R.M. & Guisan, A.
 438 (2006). Do geographic distribution, niche property and life form explain plants'
 439 vulnerability to global change? *Global Change Biology*, **12**, 1079–1093.

440 Buse, J., Schröder, B. & Assmann, T. (2007). Modelling habitat and spatial distribution of an
 441 endangered longhorn beetle—a case study for saproxylic insect conservation. *Biological*
 442 *Conservation*, **137**, 372–381.

443 Campanaro, A., Toni, I., Hadersen, S. & Grasso, D.A. (2011). Monitoring of *Lucanus cervus*
 444 by means of remains of predation (Coleoptera: Lucanidae). *Entomologia generalis*, **33**, 79–
 445 89.

446 Campanaro, A., Zapponi, L., Hadersen, S., Méndez, M., Al Fulaij, N., Audisio, P., Bardiani,
 447 M., Carpaneto, G.M., Corezzola, S., Della Rocca, F. & Harvey, D. (2016). A European
 448 monitoring protocol for the stag beetle, a saproxylic flagship species. *Insect conservation*
 449 *and diversity*, **9**, 574–584.

450 Campanaro, A., RedolfideZan, L., Hadersen, S., Antonini, G., Chiari, S., Cini, A., Mancini,
 451 E., Mosconi, F., Gasperis, S.R. de & Solano, E. (2017a). Guidelines for the monitoring of
 452 *Rosalia alpina*. *Nature Conservation*, **20**, 165.

453 Campanaro, A., Hadersen, S., Zan, L.R. de, Antonini, G., Bardiani, M., Maura, M., Maurizi,
 454 E., Mosconi, F., Zauli, A. & Bologna, M.A. (2017b). Analyses of occurrence data of
 455 protected insect species collected by citizens in Italy. *Nature Conservation*, **20**, 265.

456 Carpaneto, G.M., Baviera, C., Biscaccianti, A.B., Brandmayr, P., Mazzei, A., Mason, F.,
 457 Battistoni, A., Teofili, C., Rondinini, C. & Fattorini, S. (2015). A Red List of Italian
 458 Saproxylic Beetles: taxonomic overview, ecological features and conservation issues
 459 (Coleoptera). *Fragmenta entomologica*, **47**, 53–126.

460 Chiari, S., Carpaneto, G.M., Zauli, A., Zirpoli, G.M., Audisio, P. & Ranius, T. (2013).
 461 Dispersal patterns of a saproxylic beetle, *Osmoderma eremita*, in Mediterranean
 462 woodlands. *Insect Conservation and Diversity*, **6**, 309–318.

463 Cizek, L., Fric, Z. & Konvicka, M. (2006). Host plant defences and voltinism in European
 464 butterflies. *Ecological Entomology*, **31**, 337– 344.

465 De Marco, D.E., Montemurro, M.A. & Cannas, S.A. (2011). Comparing short and long-
 466 distance dispersal: modelling and field case studies. *Ecography*, **34**, 671–682.

467 Della Rocca, F., Bogliani, G., Breienr, F.T. & Milanesi, P. (2019). Identifying hotspots for
 468 rare species under climate change scenarios: improving saproxylic beetle conservation in
 469 Italy. *Biodiversity and Conservation*, **28**, 433–449.

470 Drag, L., Hauck, D., Pokluda, P., Zimmermann, K. & Cizek, L. (2011). Demography and
 471 dispersal ability of a threatened saproxylic beetle: a mark-recapture study of the *Rosalia*
 472 Longicorn (*Rosalia alpina*). *PLoS One*, **6**, e21345.

473 Engler, R. & Guisan, A. (2009). MigClim: predicting plant distribution and dispersal in a
 474 changing climate. *Diversity and Distributions*, **15**, 590–601.

475 Filz, K.J., Wiemers, M., Herrig, A., Weitzel, M. & Schmitt, T. (2013). A question of
 476 adaptability: Climate and habitat change lower trait diversity in butterfly communities in
 477 south-western Germany. *European Journal of Entomology*, **110**, 633–642.

478 Franklin, J. (2010). Moving beyond static species distribution models in support of
 479 conservation biogeography. *Diversity and Distributions*, **16**, 321–330.

480 Gough, L.A., Sverdrup-Thygeson, A., Milberg, P., Pilskog, H.E., Jansson, N., Jonsell, M. &
 481 Birkemoe, T. (2015). Specialists in ancient trees are more affected by climate than
 482 generalists. *Ecology and evolution*, **5**, 5632–5641.

483 Guo, F., Lenoir, J. & Bonebrake, T.C. (2018). Land-use change interacts with climate to
 484 determine elevational species redistribution. *Nature communications*, **9**, 1315.

485 Hardersen, S., Bardiani, M., Chiari, S., Maura, M., Maurizi, E., Roversi, P.F., Mason, F. &
486 Bologna, M.A. (2017). Guidelines for the monitoring of *Morimus asper funereus* and
487 *Morimus asper asper*. *Nature Conservation*, **20**, 205.

488 Harvey, D.J., Gange, A.C., Hawes, C.J. & Rink, M. (2011). Bionomics and distribution of the
489 stag beetle, *Lucanus cervus* (L.) across Europe. *Insect Conservation and Diversity*, **4**, 23–
490 38.

491 Hawes, C. J. (2008). The stag beetle *Lucanus cervus* (linnaeus, 1758) (coleoptera: lucanidae):
492 a mark-release-recapture study undertaken in one united kingdom residential garden. *Rev.*
493 *Écol. (Terre Vie)*, **63**, 139–146.

494 Higgins, S.I., Lavorel, S. & Revilla, E. (2003). Estimating plant migration rates under habitat
495 loss and fragmentation. *Oikos*, **101**, 354–366.

496 Hitch, A.T. & Leberg, P.L. (2007). Breeding distributions of North American bird species
497 moving north as a result of climate change. *Conservation Biology*, **21**, 534–539.

498 Horák, J., Chobot, K., Kohutka, A. & Gebauer, R. (2008). Possible factors influencing the
499 distribution of a threatened saproxylic beetle *Cucujus cinnaberinus* (Scopoli 1763)
500 (Coleoptera: Cucujidae). *The Coleopterists Bulletin*, **62**, 437–440.

501 Horák, J. & Chobot, K. (2009). Worldwide distribution of saproxylic beetles of the genus
502 *Cucujus* Fabricius, 1775 (Coleoptera: Cucujidae). *Saproxylic beetles—Their role and*
503 *diversity in European woodland and tree habitats*. Pensoft Publishers, Sofia–Moscow, 189–
504 206.

505 Horák, J., Vávrová, E. & Chobot, K. (2010). Habitat preferences influencing populations,
506 distribution and conservation of the endangered saproxylic beetle *Cucujus cinnaberinus*
507 (Coleoptera: Cucujidae) at the landscape level. *European Journal of Entomology*, **107**, 81–
508 88.

509 Horák, J. (2011). Response of saproxylic beetles to tree species composition in a secondary
510 urban forest area. *Urban Forestry & Urban Greening*, **10**, 213–222.

511 Kuras, T., Benes, J., Fric, Z. & Konvicka, M. (2003). Dispersal patterns of endemic alpine
 512 butterflies with contrasting popula- tion structures: *Erebia epiphron* and *E. sudetica*.
 513 *Population Ecology*, **45**, 115-123.

514 Ihlow, F., Courant, J., Secondi, J., Herrel, A., Rebelo, R., Measey, G.J., Lillo, F., de Villiers,
 515 F.A., Vogt, S. & de Busschere, C. (2016). Impacts of climate change on the global invasion
 516 potential of the African clawed frog *Xenopus laevis*. *PLoS One*, **11**, e0154869.

517 Jaeschke, A., Bittner, T., Reineking, B. & Beierkuhnlein, C. (2013). Can they keep up with
 518 climate change?—Integrating specific dispersal abilities of protected *Odonata* in species
 519 distribution modelling. *Insect Conservation and Diversity*, **6**, 93–103.

520 Jonsson, B.G., Kruys, N. & Ranius, T. (2005). Ecology of species living on dead wood—
 521 lessons for dead wood management. *Silva Fennica*, **39**, 289–309.

522 Jonsson, M., Ranius, T., Ekvall, H., Bostedt, G., Dahlberg, A., Ehnström, B., Nordén, B. &
 523 Stokland, J.N. (2006). Cost-effectiveness of silvicultural measures to increase substrate
 524 availability for red-listed wood-living organisms in Norway spruce forests. *Biological*
 525 *Conservation*, **127**, 443–462.

526 Jurc, M., Ogris, N., Pavlin, R. & Borkovic, D. (2008). Forest as a habitat of saproxylic beetles
 527 on natura 2000 sites in slovenia. *Rev. Écol.(Terre Vie)*, **63**, 61–74.

528 Lachat, T., Ecker, K., Duelli, P. & Wermelinger, B. (2013). Population trends of *Rosalia*
 529 *alpina* (L.) in Switzerland: a lasting turnaround? *Journal of Insect Conservation*, **17**, 653–
 530 662.

531 Larsson, M.C. & Svensson, G.P. (2011). Monitoring spatiotemporal variation in abundance
 532 and dispersal by a pheromone-kairomone system in the threatened saproxylic beetles
 533 *Osmoderma eremita* and *Elater ferrugineus*. *Journal of Insect Conservation*, **15**, 891– 902.

534 Macdonald, D.W. & Rushton, S. (2003). Modelling space use and dispersal of mammals in
 535 real landscapes: a tool for conservation. *Journal of Biogeography*, **30**, 607.

536 Mantyka-Pringle, C.S., Martin, T.G., Moffatt, D.B., Linke, S. & Rhodes, J.R. (2014).
537 Understanding and predicting the combined effects of climate change and land-use change
538 on freshwater macroinvertebrates and fish. *Journal of Applied Ecology*, **51**, 572–581.

539 Markovic, D., Carrizo, S., Freyhof, J., Cid, N., Lengyel, S., Scholz, M., Kasperdius, H. &
540 Darwall, W. (2014). Europe's freshwater biodiversity under climate change: distribution
541 shifts and conservation needs. *Diversity and Distributions*, **20**, 1097–1107.

542 Maurizi, E., Campanaro, A., Chiari, S., Maura, M., Mosconi, F., Sabatelli, S., Zauli, A.,
543 Audisio, P. & Carpaneto, G.M. (2017). Guidelines for the monitoring of *Osmoderma*
544 *eremita* and closely related species. *Nature Conservation*, **20**, 79.

545 Mazziotta, A., Triviño, M., Tikkanen, O.-P., Kouki, J., Strandman, H. & Mönkkönen, M.
546 (2016). Habitat associations drive species vulnerability to climate change in boreal forests.
547 *Climatic change*, **135**, 585–595.

548 Merckx, T. & Van Dyck, H. (2002). Interrelations among habitat use, behavior, and flight-
549 related morphology in two cooccurring Satyrine butterflies, *Maniola jurtina* and *Pyronia*
550 *tithonus*. *Journal of Insect Behavior*, **15**, 541– 561.

551 McCauley, S.J., Davis, C.J., Werner, E.E. & Robeson, M.S. (2014). Dispersal, niche breadth
552 and population extinction: colonization ratios predict range size in North American
553 dragonflies. *Journal of Animal Ecology*, **83**, 858–865.

554 Milanesi, P., Breiner, F.T., Puopolo, F. & Holderegger, R. (2017a). European human-
555 dominated landscapes provide ample space for the recolonization of large carnivore
556 populations under future land change scenarios. *Ecography*, **40**, 1359–1368.

557 Milanesi, P., Herrando, S., Pla, M., Villero, D. & Keller, V. (2017b). Towards continental
558 bird distribution models: environmental variables for the second European breeding bird
559 atlas and identification of priorities for further surveys. *Vogelwelt*, **137**, 53–60.

560 Milberg, P., Bergman, K., Sancak, K. & Jansson, N. (2016). Assemblages of saproxylic
561 beetles on large downed trunks of oak. *Ecology and evolution*, **6**, 1614–1625.

562 Nathan, R. (2001). Dispersal biogeography. Encyclopedia of biodiversity (ed. by S.A. Levin),
563 pp. 127-152. Academic Press, San Diego.

564 Nobis, M.P. & Normand, S. (2014). KISSMig—a simple model for R to account for limited
565 migration in analyses of species distributions. *Ecography*, **37**, 1282–1287.

566 Peterson, A.T., Soberón, J., Pearson, R.G., Anderson, R.P., Martínez-Meyer, E., Nakamura,
567 M. & Araújo, M.B. (2011). *Ecological Niches and Geographic Distributions (MPB-49)*.
568 Princeton University Press.

569 Pilskog, H.E., Birkemoe, T., Framstad, E. & Sverdrup-Thygeson, A. (2016). Effect of habitat
570 size, quality, and isolation on functional groups of beetles in hollow oaks. *Journal of Insect*
571 *Science*, **26**, 1–8.

572 Pimm, S.L., Jenkins, C.N., Abell, R., Brooks, T.M., Gittleman, J.L., Joppa, L.N., Raven, P.H.,
573 Roberts, C.M. & Sexton, J.O. (2014). The biodiversity of species and their rates of
574 extinction, distribution, and protection. *Science*, **344**, 1246752.

575 Radinger, J., Hölker, F., Horký, P., Slavík, O., Dendoncker, N. & Wolter, C. (2016).
576 Synergistic and antagonistic interactions of future land use and climate change on river fish
577 assemblages. *Global change biology*, **22**, 1505–1522.

578 Ranius, T. & Hedin, J. (2001). The dispersal rate of a beetle, *Osmoderma eremita*, living in
579 tree hollows. *Oecologia*, **126**, 363–370.

580 Ranius, T. (2002). *Osmoderma eremita* as an indicator of species richness of beetles in tree
581 hollows. *Biodiversity and Conservation*, **11**, 931–941.

582 Redolfi de Zan, L., Bardiani, M., Antonini, G., Campanaro, A., Chiari, S., Mancini, E.,
583 Maura, M., Sabatelli, S., Solano, E. & Zauli, A. (2017). Guidelines for the monitoring of
584 *Cerambyx cerdo*. *Nature Conservation*, **20**, 129.

585 Rossi de Gasperis, S., Passacantilli, C., Redolfi De Zan, L. & Carpaneto, G.M. (2016).
586 Overwintering ability and habitat preference of *Morimus asper*: a two-year mark-recapture

study with implications for conservation and forest management. *Journal of Insect Conservation*, **20**, 821–835.

Russo, D., Cistrone, L. & Garonna, A.P. (2011). Habitat selection by the highly endangered long-horned beetle *Rosalia alpina* in Southern Europe: a multiple spatial scale assessment. *Journal of Insect Conservation*, **15**, 685–693.

Sama, G. (1988). Fauna d'Italia XXVI. *Coleoptera Cerambycidae. Catalogo topografico e sinonimico*, Calderini, Bologna.

Sama, G. (2009). Fauna Europaea: *Cerambycidae. Fauna Europaea: Coleoptera. Fauna Europaea version*, **2**.

Schiegg, K. (2000). Are there saproxylic beetle species characteristic of high dead wood connectivity? *Ecography*, **23**, 579–587.

Smolik, M.G., Dullinger, S., Essl, F., Kleinbauer, I., Leitner, M., Peterseil, J., Stadler, L. & Vogl, G. (2010). Integrating species distribution models and interacting particle systems to predict the spread of an invasive alien plant. *Journal of Biogeography*, **37**, 411–422.

Soberón, J. & Peterson, A.T. (2005). Interpretation of models of fundamental ecological niches and species' distributional areas. *Biodiversity Informatics*, **2**, 1–10.

Solano, E., Mancini, E., Ciucci, P., Mason, F., Audisio, P. & Antonini, G. (2013). The EU protected taxon *Morimus funereus* Mulsant, 1862 (Coleoptera: Cerambycidae) and its western Palaearctic allies: Systematics and conservation outcomes. *Conservation Genetics*, **14** (3), 683–694.

South, A.B. & Kenward, R.E. (2001). Mate finding, dispersal distances and population growth in invading species: a spatially explicit model. *Oikos*, **95**, 53–5.

Sutherland, G.D., Harestad, A.S., Price, K. & Lertzman, K.P. (2000). Scaling of natal dispersal distances in terrestrial birds and mammals. *Conservation Ecology*, **4**, 16.

Stokland, J.N. & Siitonen, J. (2012). Mortality factors and decay succession. *Biodiversity in dead wood*. Cambridge University Press, Cambridge, 110–149.

613 Stokland, J.N., Siitonen, J. & Jonsson, B.G. (2012). Threatened saproxylic species.
614 *Biodiversity in dead wood. Cambridge University Press, Cambridge*, 356-379.

615 Stürck, J., Levers, C., van der Zanden, Emma Henriëtta, Schulp, C.J.E., Verkerk, P.J.,
616 Kuemmerle, T., Helming, J., Lotze-Campen, H., Tabeau, A. & Popp, A. (2015). Simulating
617 and delineating future land change trajectories across Europe. *Regional Environmental*
618 *Change*, 1–17.

619 Svensson, G.P., Sahlin, U., Brage, B. & Larsson, M.C. (2011). Should I stay or should I go?
620 Modelling dispersal strategies in saproxylic insects based on pheromone capture and radio
621 telemetry: a case study on the threatened hermit beetle *Osmoderma eremita*. *Biodiversity*
622 *and Conservation*, **20**, 2883–2902.

623 Thomaes, A., Kervyn, T. & Maes, D. (2008). Applying species distribution modelling for the
624 conservation of the threatened saproxylic Stag Beetle (*Lucanus cervus*). *Biological*
625 *Conservation*, **141**, 1400–1410.

626 Titeux, N., Henle, K., Mihoub, J., Regos, A., Geijzendorffer, I.R., Cramer, W., Verburg, P.H.
627 & Brotons, L. (2016). Biodiversity scenarios neglect future land-use changes. *Global*
628 *change biology*, **22**, 2505–2515.

629 Torres-Vila, L.M., Mendiola-Diaz, F.J. & Sánchez-González, Á. (2017). Dispersal differences
630 of a pest and a protected Cerambyx species (*Coleoptera: Cerambycidae*) in oak open
631 woodlands: a mark–recapture comparative study. *Ecological Entomology*, **42**, 18–32.

632 Trakhtenbrot, A., Nathan, R., Perry, G. & Richardson, D.M. (2005). The importance of long-
633 distance dispersal in biodiversity conservation. *Diversity and Distributions*, **11**, 173-181.

634 Vasudev, D., Fletcher, R.J., Goswami, V.R. & Krishnadas, M. (2015). From dispersal
635 constraints to landscape connectivity: lessons from species distribution modeling.
636 *Ecography*, **38**, 967–978.

637 Velo-Antón, G., Parra, J.L., Parra-Olea, G. & Zamudio, K.R. (2013). Tracking climate change
 638 in a dispersal-limited species: reduced spatial and genetic connectivity in a montane
 639 salamander. *Molecular ecology*, **22**, 3261–3278.

640 Walther, G.-R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T.J.C., Fromentin, J.-
 641 M., Hoegh-Guldberg, O. & Bairlein, F. (2002). Ecological responses to recent climate
 642 change. *Nature*, **416**, 389.

643 Whitmee, S. & Orme, D. (2012). Predicting dispersal distance in mammals: A trait-based
 644 approach. *Journal of Animal Ecology*, **82**, 211–221.

645 Wirth, C., Messier, C., Bergeron, Y., Frank, D. & Fankhänel, A. (2009). Old-Growth Forest
 646 Definitions: a Pragmatic View. *Old-Growth Forests*, 11–33.

647 Zuur, A.F., Ieno, E.N. & Elphick, C.S. (2010). A protocol for data exploration to avoid
 648 common statistical problems. *Methods in Ecology and Evolution*, **1**, 3–14.

649

650 BIOSKETCH

651 **Francesca Della Rocca** is an Adjunct Professor at the Department of Biology and
 652 Biotechnology, University of Pavia, and is broadly interested in the biogeography of
 653 endangered saproxylic beetles. **Pietro Milanesi** is a postdoc at the Swiss Ornithological
 654 Institute in spatial statistical modelling and is mainly interested in macroecology. Author
 655 contributions: FDR and PM conceived and designed this study, analyzed the data and wrote
 656 the first drafts of this manuscript, and alternately commented and revised the manuscript.

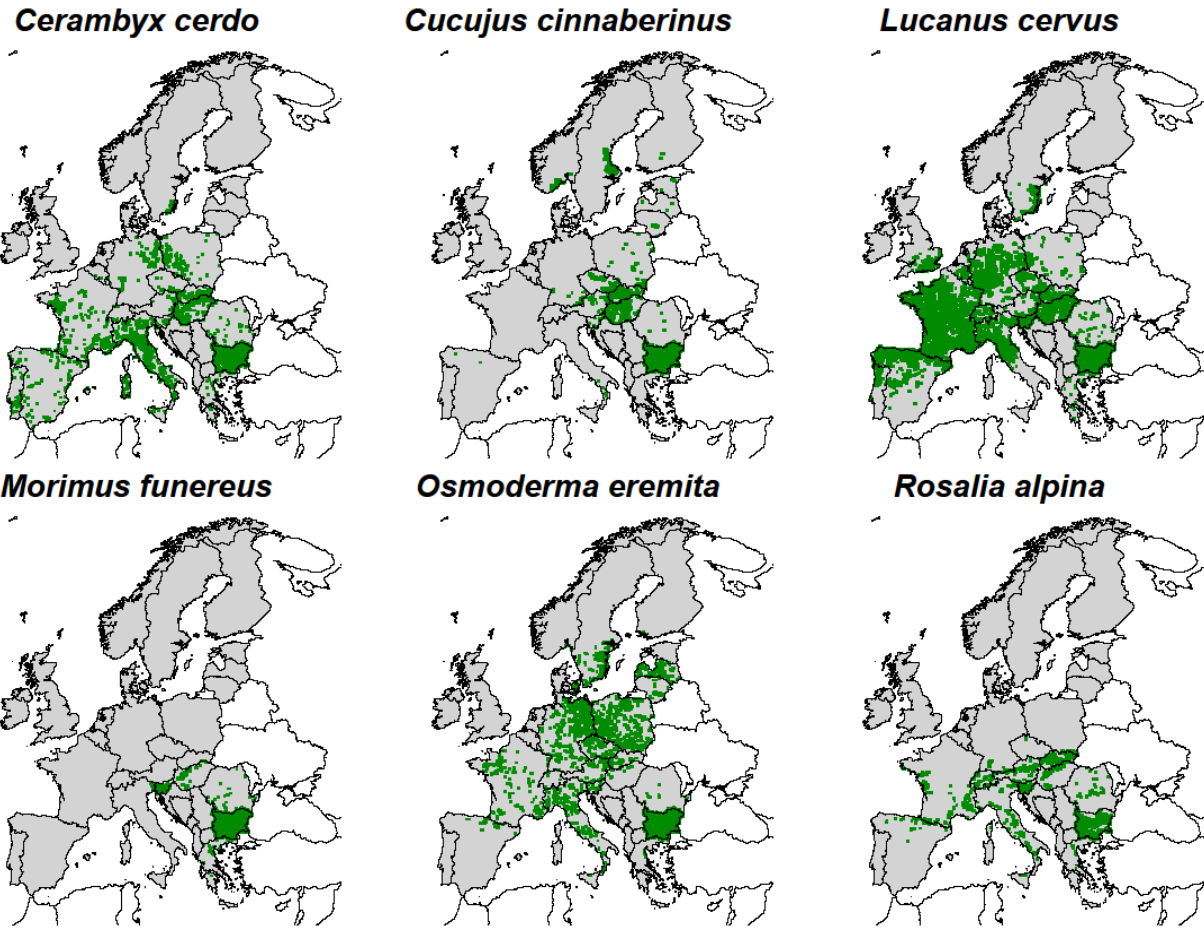
Table 1. Suitable areas (km²) for saproxylic beetles in Europe without considering species-specific dispersal distances under future (2040-50) scenarios. Four land use change scenarios: Libertarian Europe (A1), Eurosceptic Europe (A2), Social Democracy Europe (B1) and European Localism (B2), representing strong (A1), high (A2), moderate (B1) and low (B2) economic interventions and growth (Stürck *et al.*, 2015), and four Representative Concentration Pathways (RCP) derived by 11 general circulation models (GCMs), were combined to project species distribution in 2040-50. RCPs represent four possible greenhouse gas emission trajectories: low (RCP 2.6), moderate (RCP 4.5), high (RCP 6) and strong (RCP 8.5) increases in global radiative forcing (Ihlow *et al.*, 2016). Source RCPs: Intergovernmental Panel on Climate Change 5th assessment - Coupled Model Intercomparison Project Phase 5.

| Species | RCP | A1 | A2 | B1 | B2 |
|-----------------------------|-----|-----------|-----------|-----------|-----------|
| <i>Cerambyx cerdo</i> | 2.6 | 4,350,600 | 4,364,800 | 4,340,000 | 4,347,600 |
| | 4.5 | 4,393,000 | 4,408,100 | 4,385,600 | 4,383,300 |
| | 6 | 4,414,500 | 4,428,000 | 4,411,600 | 4,412,100 |
| | 8.5 | 4,423,000 | 4,421,700 | 4,415,600 | 4,406,800 |
| <i>Cucujus cinnaberinus</i> | 2.6 | 4,357,500 | 4,359,000 | 4,352,500 | 4,351,600 |
| | 4.5 | 4,370,100 | 4,383,700 | 4,378,900 | 4,377,700 |
| | 6 | 4,355,800 | 4,370,100 | 4,363,500 | 4,360,300 |
| | 8.5 | 4,437,500 | 4,445,500 | 4,441,900 | 4,440,900 |
| <i>Lucanus cervus</i> | 2.6 | 4,545,800 | 4,616,000 | 4,585,600 | 4,646,100 |
| | 4.5 | 4,532,300 | 4,608,900 | 4,558,300 | 4,624,800 |
| | 6 | 4,567,800 | 4,636,200 | 4,604,100 | 4,661,300 |
| | 8.5 | 4,508,300 | 4,577,100 | 4,539,600 | 4,601,900 |
| <i>Morimus funereus</i> | 2.6 | 3,392,800 | 3,345,200 | 3,409,900 | 3,366,800 |
| | 4.5 | 3,499,200 | 3,451,400 | 3,507,600 | 3,479,300 |
| | 6 | 3,476,700 | 3,429,100 | 3,487,700 | 3,446,400 |
| | 8.5 | 3,531,600 | 3,486,700 | 3,543,500 | 3,507,700 |
| <i>Osmoderma eremita</i> | 2.6 | 3,906,500 | 3,839,500 | 3,897,500 | 3,853,400 |
| | 4.5 | 3,879,700 | 3,817,500 | 3,864,200 | 3,814,800 |
| | 6 | 3,852,700 | 3,786,500 | 3,832,600 | 3,784,900 |
| | 8.5 | 3,880,500 | 3,817,500 | 3,862,700 | 3,822,000 |
| <i>Rosalia alpina</i> | 2.6 | 3,858,000 | 3,903,100 | 3,891,900 | 3,923,500 |
| | 4.5 | 3,867,900 | 3,921,000 | 3,906,400 | 3,945,200 |
| | 6 | 3,916,400 | 3,974,200 | 3,974,500 | 3,996,100 |
| | 8.5 | 3,923,200 | 3,975,100 | 3,966,900 | 3,996,900 |

Table 2. Suitable areas (km²) for saproxylic beetles in Europe accounting for species-specific dispersal distances under future (2040-50) scenarios. Four land use change scenarios: Libertarian Europe (A1), Eurosceptic Europe (A2), Social Democracy Europe (B1) and European Localism (B2), representing strong (A1), high (A2), moderate (B1) and low (B2) economic interventions and growth (Stürck *et al.*, 2015), and four Representative Concentration Pathways (RCP) derived by 11 general circulation models (GCMs), were combined to project species distribution in 2040-50. RCPs represent four possible greenhouse gas emission trajectories: low (RCP 2.6), moderate (RCP 4.5), high (RCP 6) and strong (RCP 8.5) increases in global radiative forcing (Ihlow *et al.*, 2016). Source RCPs: Intergovernmental Panel on Climate Change 5th assessment - Coupled Model Intercomparison Project Phase 5.

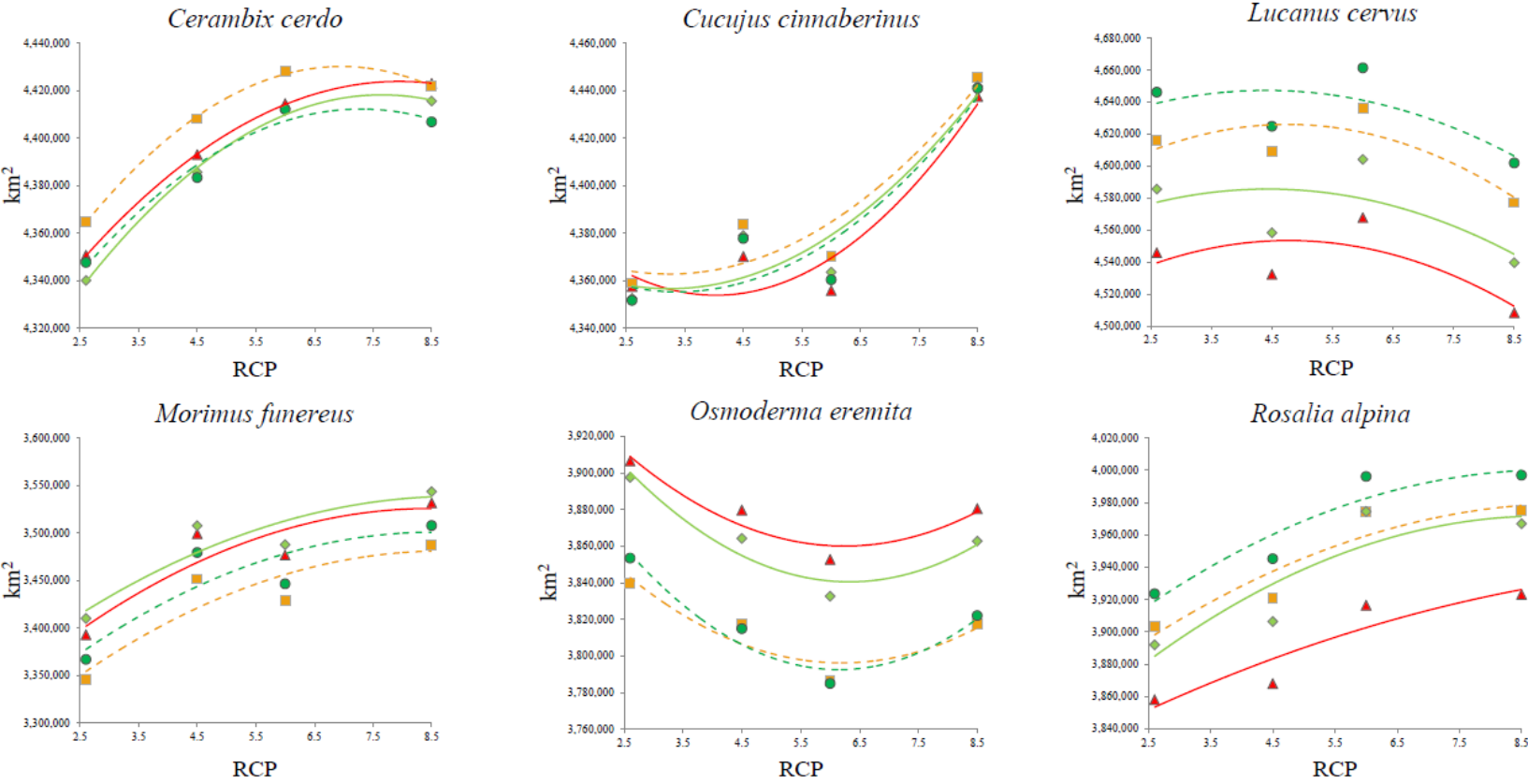
| Species | RCP | A1 | A2 | B1 | B2 |
|-----------------------------|-----|-----------|-----------|-----------|-----------|
| <i>Cerambyx cerdo</i> | 2.6 | 1,079,600 | 1,072,100 | 1,081,300 | 1,074,500 |
| | 4.5 | 1,083,500 | 1,083,000 | 1,087,200 | 1,076,100 |
| | 6 | 1,078,800 | 1,081,600 | 1,086,100 | 1,073,700 |
| | 8.5 | 1,090,000 | 1,089,700 | 1,091,500 | 1,086,500 |
| <i>Cucujus cinnaberinus</i> | 2.6 | 573,900 | 571,500 | 571,100 | 574,300 |
| | 4.5 | 578,300 | 577,700 | 578,100 | 583,200 |
| | 6 | 578,600 | 576,400 | 578,000 | 580,200 |
| | 8.5 | 583,600 | 582,200 | 584,200 | 585,600 |
| <i>Lucanus cervus</i> | 2.6 | 1,728,200 | 1,704,100 | 1,728,200 | 1,713,300 |
| | 4.5 | 1,747,700 | 1,723,700 | 1,747,500 | 1,729,900 |
| | 6 | 1,738,300 | 1,708,100 | 1,735,000 | 1,719,400 |
| | 8.5 | 1,757,500 | 1,727,100 | 1,749,500 | 1,737,900 |
| <i>Morimus funereus</i> | 2.6 | 179,000 | 179,300 | 182,800 | 173,900 |
| | 4.5 | 177,600 | 180,000 | 181,400 | 175,600 |
| | 6 | 179,300 | 181,500 | 183,000 | 176,900 |
| | 8.5 | 178,700 | 181,100 | 183,200 | 177,400 |
| <i>Osmoderma eremita</i> | 2.6 | 1,041,000 | 1,029,800 | 1,032,300 | 1,030,900 |
| | 4.5 | 1,021,300 | 1,015,000 | 1,022,900 | 1,018,500 |
| | 6 | 1,032,400 | 1,016,600 | 1,020,300 | 1,017,300 |
| | 8.5 | 1,018,200 | 1,005,700 | 1,009,900 | 1,009,700 |
| <i>Rosalia alpina</i> | 2.6 | 566,000 | 564,200 | 561,700 | 570,000 |
| | 4.5 | 574,800 | 570,900 | 569,600 | 579,300 |
| | 6 | 572,400 | 565,800 | 567,600 | 571,900 |
| | 8.5 | 579,100 | 571,400 | 570,900 | 581,300 |

679 Figure 1. Study area (in grey) and current species ranges, according to European Environment Information and Observation Network (EIONET)
680 Central Data Repository server, Swiss Biological Records Center and Norwegian Taxonomy Initiative, of the six saproxylic species considered (in
681 green).



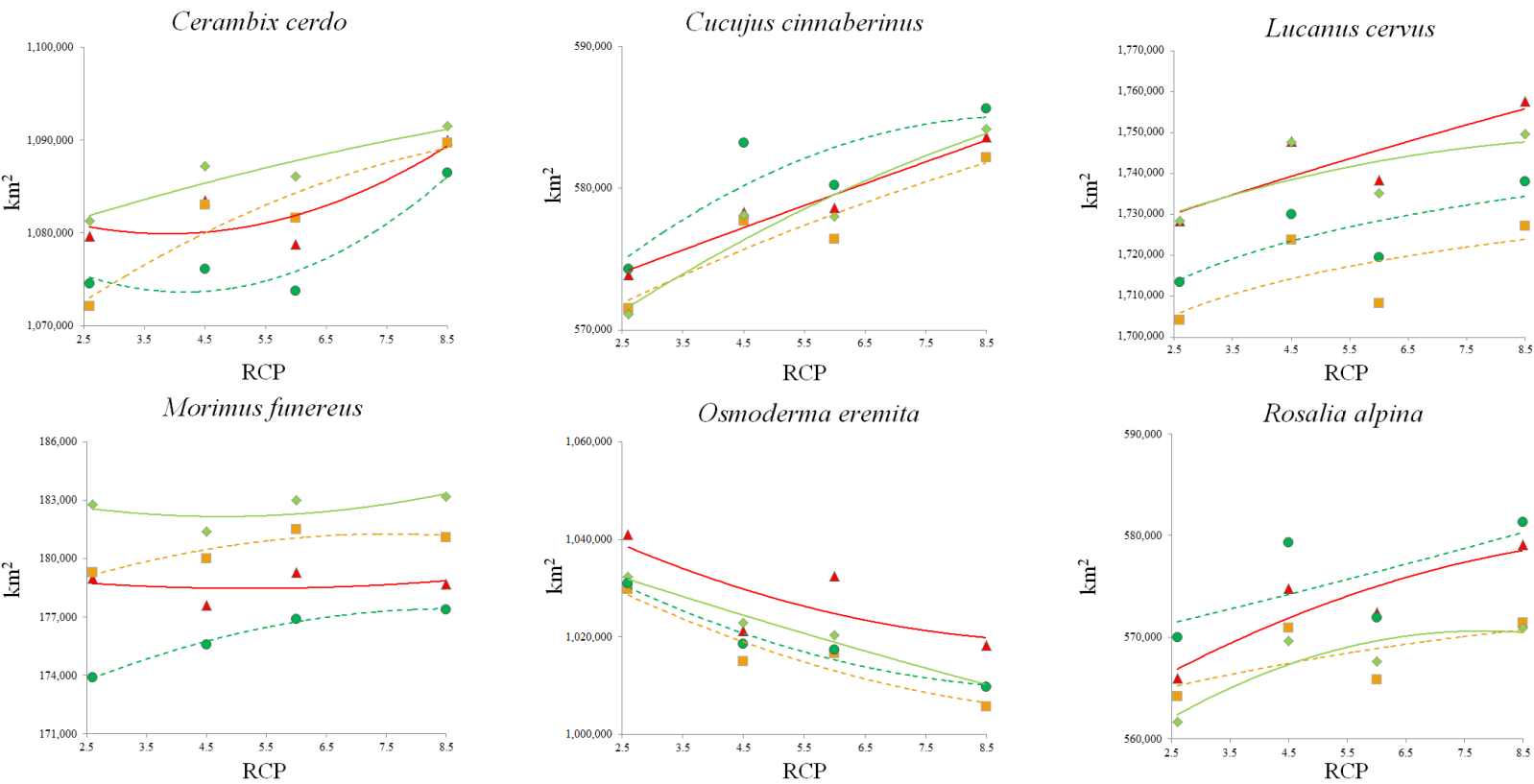
682

683 Figure 2. Response curves of suitable areas (km²), generated without accounting for saproxylic beetles dispersal distances, in relation to four land
 684 use change scenarios (Stürck *et al.*, 2015) and four Representative Concentration Pathways (RCP) derived by 11 general circulation models (GCMs)
 685 for the period 2040-50 in Europe. Filled red for A1, dashed orange for A2, filled light green for B1 and dashed dark green for B2. Source RCPs:
 686 Intergovernmental Panel on Climate Change 5th assessment - Coupled Model Intercomparison Project Phase 5.



687

688 Figure 3. Response curves of suitable areas (km²), accounting for saproxylic beetles dispersal distances, in relation to four land use change scenarios
 689 (Stürck *et al.*, 2015) and four Representative Concentration Pathways (RCP) derived by 11 general circulation models (GCMs) for the period 2040-
 690 50 in Europe. Filled red for A1, dashed orange for A2, filled light green for B1 and dashed dark green for B2. Source RCPs: Intergovernmental
 691 Panel on Climate Change 5th assessment - Coupled Model Intercomparison Project Phase 5.



692

693 **SUPPORTING INFORMATION**

694 Additional supporting information may be found online, in the Supporting Information
695 section at the end of the article.