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Potential range of impact of an ecological trap network: the case of timber stacks and the *Rosalia longicorn*

Paweł Adamski¹ · Jakub Michalcewicz² · Adam Bohdan³ · Michał Ciach⁴

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Abstract

Although the negative impact of timber stacks on populations of saproxylic beetles is a well-known phenomenon, there is relatively little data concerning the scale of this impact and its spatial aspect. Beech timber stored in the vicinity of the forest can act as an ecological trap for the *Rosalia longicorn* (*Rosalia alpina*), so in this study we have attempted to determine the spatial range of the impact of a network of timber stacks. Timber stacks in the species' range in the study area were listed and monitored during the adult emergence period in 2014–2016. Based on published data relating to the species' dispersal capabilities, buffers of four radii (500, 1000, 1600, 3000 m) were delineated around the stacks and the calculated ranges of potential impact. The results show that the percentage of currently known localities of the *Rosalia longicorn* impacted by stacks varies from 19.7 to 81.6%, depending on the assumed impact radius. The percentage of forest influenced by timber stacks was 77% for the largest-radius buffer. The overall impact of the ecological trap network is accelerated by fragmentation of the impact-free area. It was also found that forests situated close to the timber stacks where the *Rosalia longicorn* was recorded were older and more homogeneous in age and species composition than those around stacks where the species was absent. Such results suggest that timber stacks act as an ecological trap in the source area of the local population.

Keywords Spatio-temporal population dynamics · Timber stack · Ecological trap · Forest management · Species conservation · Dispersal behaviour

Introduction

The spatial approach is widely used in conservation biology, both in theoretical studies and for supporting decision-making in conservation actions. The spatio-temporal approach, the core of the metapopulation theory (Gilpin and Hanski 1991; Hanski et al. 1996; Schnell et al. 2013; Porter and Ellis 2011), primarily addresses the relation between habitat structure and population dynamics (Gering et al. 2003; Hughes et al. 2000; Bulman et al. 2007). It enables a more reliable description of a population's decline, and the subsequent design of effective conservation measures (Harrison 1991; Adamski and Witkowski 2007; Cormont et al. 2011; Hanski 2011). In conservation practice, however, application of the metapopulation model is often limited by gaps in the knowledge of both population dynamics parameters and the habitat requirements of species (Cushman 2006; Rudnick et al. 2012). Despite its obvious limitations, expert knowledge can also be quite effective in nature conservation when urgent decisions have to be made (Kuhnert 2011; Martin et al. 2012).

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One of the critical aspects of conservation biology is the spatial pattern of sources of threat; knowledge of this pattern is indispensable for any assessment of their overall impact (Coffin 2007; Nelson et al. 2009; Groot et al. 2010). A key, spatially-related threat at the population level is the ecological trap (Dwernychuk and Boag 1972; Oaks et al. 2004). This is the situation when an organism—usually an animal—attempts to reproduce in sites where reproduction is in fact not possible, even though such sites are plentiful (Kokko and Sutherland 2001; Gilroy and Sutherland 2007; Robertson et al. 2010). It is usually the effect of a broken link between the real quality of a site and the aspect that persuades an organism to select it (Oaks et al. 2004), or of perceptual errors as in the mechanism of sensory exploitation (Horváth et al. 2007; Norris et al. 2013). Ecological traps are usually fairly easily distinguishable, often anthropogenic, objects in space (Robertson and Hutto 2006; Horváth et al. 2007; Harabiš and Dolný 2012), so analysing the extent and range of their impact is perfectly feasible. A well-known example of this approach is analysing the spatial impact of light pollution: there are convincing descriptions of how artificial light sources act as ecological traps (Longcore and Rich 2004; Robertson et al. 2013). There are also numerous studies highlighting the importance of the spatial pattern of ecological traps (Weldon and Haddad 2005; Kempnaers et al. 2010; Bates et al. 2014). The presence of ecological traps in an area inhabited by a population causes the quality of the local habitat to deteriorate, even though other parameters, such as the availability of critical resources, are maintained. As habitat quality is considered to be the key factor responsible for the success of conservation efforts related to insect populations (Adamski and Witkowski 2007; Sutherland et al. 2013), it is important to recognise whether it is impoverished by ecological traps.

Studies of light pollution have demonstrated the significance of the spatial pattern of threat factors, which is crucial to assessing their overall effect (Longcore and Rich 2004; Gaston et al. 2012). In our study, we refer to the spatial context of a well-known type of ecological trap, i.e. timber stacks, which endangers xylophagous insects (Jonsell et al. 2007; Hedin et al. 2008; Jonsell 2008; Lassauce et al. 2012; Adamski et al. 2013, 2016). Although attention has been drawn to the negative impact of timber stacks on saproxylophagous beetle populations for quite a long time (Gutowski 2004b; Jonsell et al. 2007; Hedin et al. 2008; Jonsell 2008; Michalcewicz et al. 2011), research has focused mainly on determining the species composition of saprophages colonising potential traps (Jonsell 2008) or on the parameters of the stored timber preferred by these organisms (Jonsell et al. 2007; Hedin et al. 2008; Lassauce et al. 2012), without reference to the spatial range of this impact.

One endangered species threatened by this forest management practice is the Rosalia longicorn *Rosalia alpina* (L.)

(Cerambycidae) (IUCN 2014). Timber stacks are known to be a serious danger to the Rosalia longicorn: this has been highlighted in numerous publications and species management plans (e.g. Starzyk 2004; Witkowski 2007; Horák et al. 2010; Nieto and Alexander 2010; Michalcewicz and Ciach 2012), and explicitly demonstrated in studies undertaken in the Carpathians (Adamski et al. 2013, 2016). However, no attempts have been made to standardise the methodology for assessing the overall impact of a network of timber stacks on populations of xylophages, including the Rosalia longicorn. The present work aimed to assess the spatial extent of the potential impact of a timber stack network on a Rosalia longicorn population. We applied a GIS-based method, which takes into consideration a species' movement capabilities, in order to assess the potential spatial impact of ecological traps.

Materials and methods

Species

The Rosalia longicorn has a Euro-Caucasian distribution, and its geographical range covers central and southern Europe (Sama 2002; Shapovalov 2012; IUCN 2014). However, its current central European distribution of this species is highly fragmented, and it inhabits only a few isolated areas in its range (Binner and Bussler 2006). In this part of Europe, the Rosalia longicorn occurs mainly in mature, > 100-year-old open canopy, mountain forest stands of European beech *Fagus sylvatica* (Burakowski et al. 1990). Classified by some authors as a primeval forest relict (Gutowski 2004a; Pawłowski 2008; Eckelt et al. 2017), the species is commonly treated as an icon of saproxylic biodiversity throughout its range.

At the microhabitat scale, this beetle inhabits old, strongly insolated trees that have been damaged by frost, wind or fire, yet are still alive. It also inhabits dead trees, both standing and fallen, and freshly logged or stockpiled timber. It occurs in both softened wood, rotten and strongly decomposed by wood-decay fungi, and in the bark-less yet still hard and undecayed wood of live trees (Capecki 1969; Gutowski 2004b; Starzyk 2004). Regardless of the extent and the stage of decay, strong insolation is a feature common to the species' breeding substrate (Russo et al. 2011). The Rosalia longicorn is highly mobile and its movement ability has been assessed at between 3000 m (Drag et al. 2011) to more than 10 km (Lachat et al. 2013).

Study area

The study was conducted in the Polish part of the western Bieszczady Mountains (Carpathians, SE Poland)

(Kondracki 2011). The Bieszczady are part of the eastern Beskids, which stretch from the Łupków Pass and the valley of the River Osława in the west to the Uzh Pass and the sources of the River San on the Ukrainian border in the east (Fig. 1). A characteristic feature of the western Bieszczady is the lack of upper alpine Norway spruce *Picea abies* forests, as a result of which deciduous forests are dominant in the area. The Bieszczady National Park was founded in 1973 and currently covers an area of 270.6 km². Subsequently, two landscape parks—the Cisna-Wetlina and San Valley LP—were established in 1992. All three parks were included in the international biosphere reserve delineated on the borders of Poland, Slovakia and Ukraine. There are numerous nature reserves in this region (Kondracki 2011). The PLC180001 Bieszczady Natura 2000 area covers much of the western Bieszczady Mts.

The territory in question is therefore covered by several, partially overlapping, protected areas with varying conservation regimes. The National Park and a few small natural reserves are therefore strictly protected, whereas in the Landscape Park and Natura 2000 sites, the activities of the forest industry are restricted only by the provisions of the management plans of protected areas, which are usually focused on particular species, habitats or other natural values (Parliament of the Republic of Poland 2004). The average annual amount of beech timber logged in 2012–2015 was estimated at 237.2 m³/km², and the harvested timber was stacked in fixed locations, designated by the forest administration (Adamski et al. 2016).

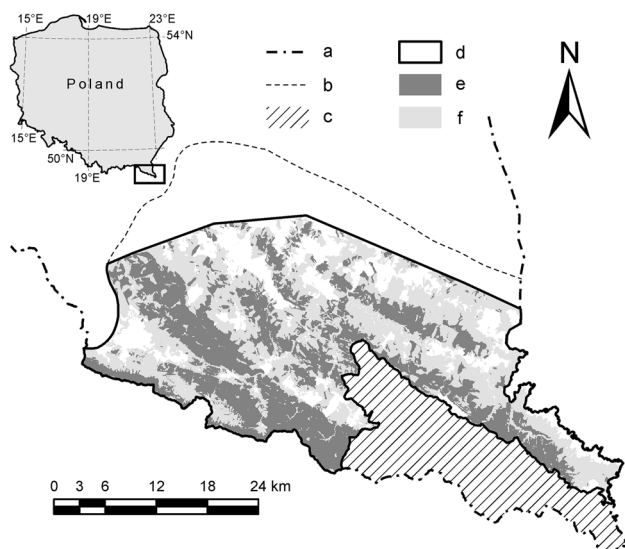


Fig. 1 The study area: (a) Polish state border, (b) border of the Western Carpathians region [according to Kondracki (2011)], (c) Bieszczady National Park, (d) study area, (e) forest with dominant beech, (f) forest with dominance of another tree species

Data collection and analysis

This survey of timber stacks was carried out in the part of Bieszczady Mts. covered by the currently known distribution range of the *Rosalia longicorn* (Michalcewicz and Ciach 2015); the Bieszczady National Park, where timber is neither harvested nor stored, was excluded from the analysis. In 2014–2016 we searched the study area for timber stacks and then monitored them during the adult emergence period (July–August). The locations of the beech timber stacks were recorded using a GPS data logger. Each stack was monitored and the presence of the species noted.

Buffer zones for assessing the potential impact of timber stacks on the *Rosalia longicorn* population were delimited around each one. Buffer zone radii, corresponding to the dispersal potential categories of imagines, were determined on the basis of data relating to species movement (Fig. 2): (A) 500 m—the dispersal range of approx. 8% of individuals; (B) 1000 m—the dispersal range of approx. 4% of individuals; (C) 1600 m—the maximum dispersal range found during a mark-capture-recapture trial; and (D) 3000 m—potential dispersal range based on modelling (Drag et al. 2011).

If the buffers surrounding two or more stacks partially overlapped, they were combined into a common buffer. In order to avoid methodological artefacts, the parts of buffers extending beyond the predetermined study area were excluded from the analysis (Sadahiro 2005). The area of forest covered by buffers of different radii was calculated separately for (1) the whole forest-covered area and (2) forest with dominant beech only. Moreover, the numbers of currently known localities of the *Rosalia longicorn* (Michalcewicz and Ciach 2015) covered by the buffers were counted. These numbers were recalculated to the proportions with the confidence calculated according to Newcombe (1998).

Based on the percentage of the total forest area covered by buffers, the expected number of known *Rosalia longicorn* localities was calculated using the formula:

$$N_e = N_t \frac{A_b}{A_t} \quad (1)$$

where N_e is expected number of localities, N_t is total number of localities, A_b is forest area covered by buffers, A_t is total forest area at the study area.

Calculations were done for each buffer radius, separately for the buffers around the stacks where the *Rosalia longicorn* had been observed and for those where the species had not been observed, as well as the whole area covered by the buffers. Subsequently, the expected numbers were compared with the observed ones using the chi-squared (χ^2) test.

The forests beyond the buffers were treated as impact-free stands. The distance to the nearest neighbour was designated for each buffer radius in such areas. Then, the relation

Fig. 2 Areas free from the impact of the network of ecological traps, depending on the adopted radius of impact (based on the movement capabilities of the Rosalia longicorn *Rosalia alpina* [Drag et al. (2011)]. **A** 500 m buffers, **B** 1000 m buffers, **C** 1600 m buffers, **D** 3000 m buffers; (a) Polish state border, (b) border of the study area, (c) Bieszczady National Park, (d) beech timber stacks, (e) forest with dominant beech, (f) forest with dominance of another tree species

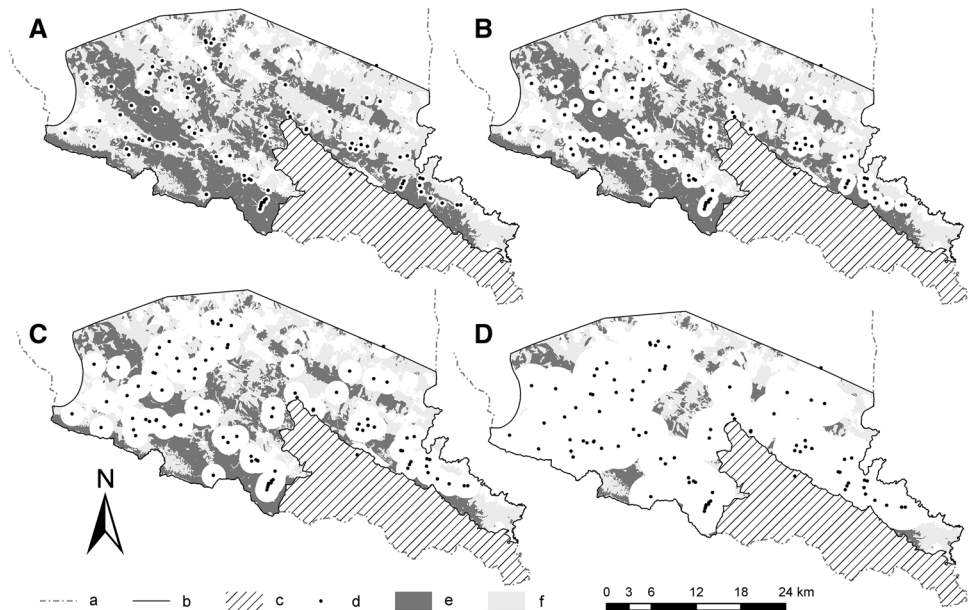


Table 1 Average size of forest units within the study region, Bieszczady Mts. (Carpathians, SE Poland)

	Total	Dominant species: European beech <i>Fagus sylvatica</i>	Dominant species: other
Number	8470	2838	5632
Mean area [ha]	9.67	15.17	7.34
Area SD	12.30	14.60	10.34
Area median	4.47	10.03	3.26
Lower quartile	1.56	3.69	1.20
Upper quartile	12.99	23.48	8.87

As the distribution of forest unit areas differs significantly from the normal ($W=0.3268$; $p<0.0001$), the mean and SD are less useful parameters than medians and quartiles

between this distance and the buffer radius was analysed. As the distribution of the nearest distance between intact impact-free forest areas deviated significantly from normality (Shapiro–Wilk test; $W_{726}=0.120$, $p<0.0001$, Table 1), the non-parametric Kruskal–Wallis test and the Steel–Dwass post-hoc test were applied.

Data on forest type were provided by the Forest Data Bank (FDB 2016). This contains maps and descriptive data relating to forest areas managed by the State Forests National Forest Holding as of 1 January 2016, including information on the dominant tree species as well as the stand size and age of each forest unit (forest management practice in Poland divides all forests into spatial units, where forests are relatively uniform with respect to species composition, age, height and spatial structure).

To determine the parameters of forest stands related to the presence of the *Rosalia longicorn*, the buffers around the

stacks with and without records of the species (Fig. 2) were compared for each buffer radius separately, using a logistic regression model for the binominal distribution of the effect and the logit linked function. The dependent variable was the absence or presence of the *Rosalia longicorn* on timber stacks, whereas predictors were forest stand parameters related to the known habitat requirements of the *Rosalia longicorn*: (i) forest stand age and (ii) dominance of beech were used as indicators of habitat quality, since the *Rosalia longicorn* is associated with mature beech forests; (iii) forest unit size, which is considered an indicator of habitat heterogeneity. Large units implied contiguous patches of homogeneous forest stand, whereas small ones were treated as indicators of habitat heterogeneity, since they contain small patches of stands of different ages and dominant species. Stand age and forest unit area were expressed as continuous variables, whereas beech dominance was a categorical one.

ArcGIS 10.1 (ESRI 2010) and the *mlogit* package (Croissant 2015) developed in R software (R Core Development Team 2015) were used in the analysis.

Results

The majority (77.6%) of the study area is covered with forest stands (total area = 998.29 km²), and European beech is the dominant tree species in 35.4% of the forest units ($N=8015$). There was, however, a significant difference (Wilcoxon test; $Z=28.90$, $p<0.0001$) between the average sizes of the forest units with dominant beech and of those where other tree species were dominant (Table 2). As a result, forest units with dominant beech constituted 46.46% (total area = 775.06 km²) of all forest stands.

Table 2 Areas of buffer zones and areas of tree stands within them—these are the actual areas and percentages of the same type of forest within the study region

Buffer zone radius (m)	All forests		Forest with dominant beech		Forest with dominance of another species	
	Area [km ²]	Ratio [%]	Area [km ²]	Ratio [%]	Area [km ²]	Ratio [%]
500	50.55	6.5	21.79	6.0	28.75	6.9
1000	163.63	21.1	77.44	21.4	86.20	20.8
1600	328.17	42.3	160.88	44.5	167.29	40.4
3000	599.67	77.4	301.77	84.5	297.90	72.0

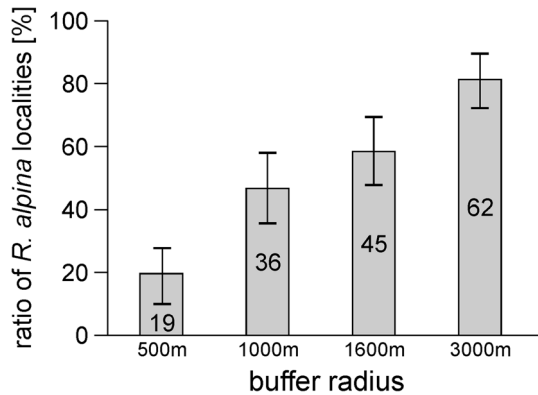


Fig. 3 Ratios (±CL) of *Rosalia longicorn* *Rosalia alpina* localities covered by buffers with different radii

A total of 88 sites (0.88/10 km² of the total area and 1.14/10 km² of the forest area) where beech timber was stored during the *Rosalia longicorn*'s emergence period were located in the study area. The proportion of forest area covered by the buffers delineated around the timber stacks varied from 6.5% in the buffers of 500 m radius to 77.4% in the 3000 m buffers (Fig. 2; Table 2). Owing to the high density of timber stacks and buffer overlap, forest areas situated beyond the range of the potential impact of ecological traps form scattered and isolated patches of habitats (Figs. 2, 3).

The buffers covered a significant proportion of the localities where the *Rosalia longicorn* is currently recorded (Fig. 3).

The distance between compact, impact-free forest patches depended on the buffer radius (Kruskal–Wallis test; $H = 15.631$, $p = 0.004$) and was significantly higher for the 3000 m buffers than for the 500 and 1000 m buffers (Fig. 4).

Adult *Rosalia longicorn*s were recorded at 15% ($N = 88$) of the monitored timber stacks. Comparison of forest traits around the timber stacks where *Rosalia longicorn*s were or were not present indicated that within the 500 m radius, the forest stands surrounding a stack with *Rosalia longicorn*s were older than those around stacks without records of this beetle (Table 3). For larger buffer radii, stand age lost its significance in favour of forest unit area (Table 3): the forest units surrounding timber stacks with *Rosalia longicorn*s were larger than those where the species was not recorded.

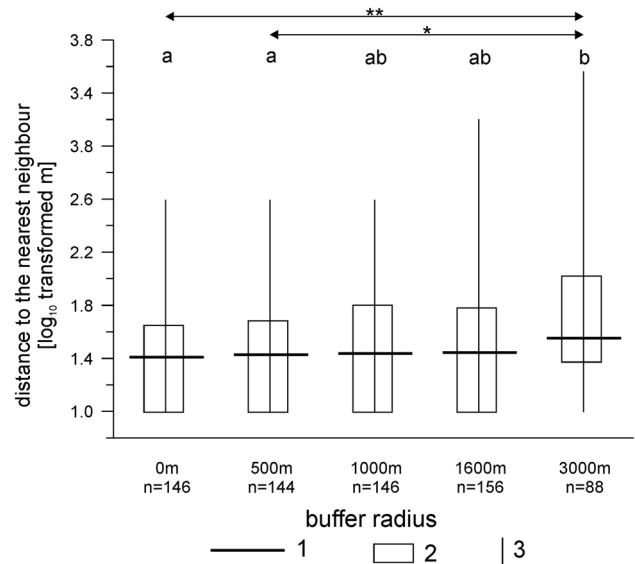


Fig. 4 Distances to the nearest impact-free forest area; 1—median, 2—lower/upper quartile, 3—range, 4—differences obtained from post hoc comparison; * $p < 0.05$, ** $p < 0.001$

The numbers of known localities of the *Rosalia longicorn* covered by the buffers surrounding the stacks significantly exceeded the expected values (calculated using formula 1). Moreover, the number of previously recorded localities was significantly higher in the buffers around stacks where the species was observed during the study period than in the buffers surrounding the stacks where the *Rosalia longicorn* was not present (Table 4).

Discussion

Timber stacks in the Bieszczady Mts., known to be a threat to the *Rosalia longicorn* (Gutowski 2004b), fulfil the criteria of an ecological trap for this species (Adamski et al. 2013, 2016). However, no study investigating the importance of their spatial pattern has been carried out to date.

The general mechanism of the impact of timber stacks on saproxylic insects is similar to that relating to artificial light sources, i.e. imagines are attracted to sites of incorrectly assessed habitat quality, where reproduction

Table 3 Logistic regression models comparing parameters of forest stands (age, area and European beech *Fagus sylvatica* dominance) surrounding the stacks where the Rosalia longicorn *Rosalia alpina* was present or absent; species presence/absence was used as a dichotomous dependent variable (1/0) and four separate models for various radii of buffer zones (500, 1000, 1600 and 3000 m) are presented

Parameter	Estimate	SE	t	p
Buffer 500 m ($F_{3, 1082} = 6.341$; $p = 0.0001$)				
Intercept	0.726	0.300	2.419	0.016
Age	0.011	0.003	3.728	0.000
Area	0.017	0.016	1.047	0.295
Beech dominance	0.151	0.214	0.705	0.481
Buffer 1000 m ($F_{3, 2312} = 4.364$; $p = 0.0052$)				
Intercept	1.015	0.183	5.547	0.001
Age	0.003	0.002	1.811	0.070
Area	0.018	0.007	2.643	0.008
Beech dominance	0.121	0.127	0.954	0.340
Buffer 1600 m ($F_{3, 4083} = 5.531$; $p = 0.0000$)				
Intercept	1.018	0.128	7.932	0.000
Age	0.002	0.001	1.665	0.096
Area	0.013	0.004	3.176	0.002
Beech dominance	0.087	0.088	0.985	0.324
Buffer 3000 m ($F_{3, 6256} = 5.298$; $p = 0.0014$)				
Intercept	1.020	0.083	12.318	0.000
Age	-0.002	0.001	-0.250	0.803
Area	0.009	0.002	3.887	0.000
Beech dominance	0.037	0.057	0.511	0.511

is impossible or ineffective. Unfortunately, however, the mechanisms by which dead wood attracts saproxylic beetles are more complex, mostly because the propagation of the chemical signal depends on numerous local conditions,

such as weather as well as micro- and mesoclimatic parameters (Hedgren 2007; Svensson and Larsson 2008; Chiari et al. 2013). Moreover, recent studies have demonstrated the presence of an aggregating pheromone produced by the Rosalia longicorn (Žunic-Kosi et al. 2017). This complicates analyses of the extent of timber stack impact on saproxylic insects, because the attractive power of the timber stack seems to depend not only on the presence of large amounts of the preferred host plant (potential breeding substrate), but also on the presence of Rosalia longicorn individuals themselves, which may reinforce the luring effect of timber stacks. Consequently, the present interpretation of the impact of an ecological trap should be treated as a minimum estimate. There are several reasons for this. The first relates to the radii of this impact. As mentioned earlier, these were taken from the study by Drag et al. (2011). One may argue that less than 4% of individuals disperse over distances greater than 1000 m. However, this proportion applies only to recaptured individuals. In the discussion of their results, those authors admitted that, because of the rather low level of detectability, the ratios and dispersal distances they calculated were underestimated (see Drag et al. 2011). They suggested that the maximum migration distance of Rosalia longicorns could be as far as 10 km (Drag et al. 2011), a statement corroborated in another study (Lachat et al. 2013). Another simplification made during the present work was to treat timber stacks as points, whereas in fact they may cover areas of several hundred square metres or extend for some considerable distance, e.g. along forest roads. For this reason, the actual area of an ecological trap's impact range is larger than that derived from point locations. Moreover, our analyses did not take into account the small firewood

Table 4 Numbers of the known localities of *Rosalia alpina* in the study area covered by buffer with different radii

	In buffers around the stacks		Outside the buffers C	Chi squared for comparison	
	RA was recorded A	RA was not recorded B		(A + B) vs. C	A vs. B
500 m					
Observed	1	4	71	$\chi^2 = 13.754$	$\chi^2 = 10.467$
Expected	7	12	57	$p = 0.001$	$p = 0.005$
1000 m					
Observed	3	13	60	$\chi^2 = 21.111$	$\chi^2 = 12.648$
Expected	15	21	40	$p < 0.001$	$p = 0.002$
1600 m					
Observed	7	25	44	$\chi^2 = 9.207$	$\chi^2 = 9.375$
Expected	21	24	31	$p = 0.010$	$p = 0.009$
3000 m					
Observed	17	42	17	$\chi^2 = 0.788$	$\chi^2 = 28.106$
Expected	39	23	14	$p = 0.674$	$p < 0.001$

The expected number was calculated using formula 1, based on the proportion of the study area (Table 2)

storage sites situated near properties situated in the study area, which could also be a part of the ecological trap network affecting the *Rosalia longicorn* population.

One can make the point that such limited and imprecise information about the species' dispersal range precludes reliable analyses of the spatial scale of ecological trap impact. However, as mentioned earlier, conservation practice shows that for practical purposes, expert knowledge—even if quite imprecise—is often effective and therefore useful (Kuhnert 2011; Martin et al. 2012). Therefore, despite these limitations, the proposed method seems to provide a convincing approximation of the potential range of impact of timber stacks on the *Rosalia longicorn*. The results indicate that just ca. 23% of the forests in the study area are beyond the reach of the impact of timber stacks (Table 2; Fig. 2). One could argue that the estimated area of impact for smaller buffer radii is much lower. However, a number of threat factors can lead to habitat perforation—a particular instance of fragmentation, when spots of hostile habitat appear within patches of suitable habitat (Miller et al. 2001); this can influence large-scale habitat quality (Forsman et al. 2010). It is also worth stressing that such impact-free forest does not cover a compact area but is fragmented and distributed across the whole study area, and the distances between them increase with the buffer radii (Figs. 2, 4). Comparison of the spatial distributions of the timber stacks and previously recorded sites of the *Rosalia longicorn* shows that they are related. Moreover, the known localities more often occur in the vicinity of the stacks where the *Rosalia longicorn* was observed (Table 4; Fig. 5). This shows that timber stacks, effectively as ecological traps, are in operation in the core of the local population. Unfortunately, there are insufficient data for estimating the strength of this impact, although the above-mentioned co-occurrence itself is an important sign of the threat. It should, however, be emphasised that the

core area of the distribution of a population is usually also its source (Pulliam 1988). Therefore, the whole population is extremely sensitive to factors reducing reproductive effectiveness in such an area (Holt 1996; Lawton 1993; Diaz 1996). As a result, the decrease in reproductive success in the population's source area, such as a reduction of density, may shift to a population sink (Holt 1985; Dias 1996), so there is a risk of the threat being underestimated. In the case of the study area, one may speculate that the difference between the spatial distributions of the *Rosalia longicorn* localities in the study area in 2000–2013 (Michalcewicz and Ciach 2015) and 2014–2016 (Fig. 5) may be explained by the above mechanism.

The source area of the population can also be determined by comparing habitat parameters—here, by comparing the surroundings of stacks occupied by the *Rosalia longicorn* and of those where this species was not present. The comparison showed that the forest near the timber stacks occupied by the *Rosalia longicorn* demonstrated the importance of forest stand homogeneity. The main factor governing the presence of the *Rosalia longicorn* is the surface area of a forest unit, whereas the dominance of European beech is insignificant and the age of a tree stand is important only in the immediate vicinity of the stack (500 m buffer). As beech is the *Rosalia longicorn*'s main host plant, the lack of a significant relationship between its dominance in the vicinity of a timber stack and the species' presence there may seem rather surprising. However, it should be stressed that European beech occurs as an admixture in the majority of forest stands in which it is not the dominant species (Kozak et al. 2002; Jaworski and Kołodziej 2004). According to Forest Management Plans of the study area, 85% of the forest units in Bieszczady Mts. contain at least 10% European beech (RDLP Krosno 2015a). This prevalence of European beech suggests that most of the forest stands in the study area constitute potential habitat for the *Rosalia longicorn*. In consequence, this beetle may find trees suitable for larval development in almost all types of forest stands. The significant part played by forest unit area can also be explained by the relation between forest heterogeneity and the degree of habitat fragmentation: larger forest units that are uniform habitat patches may favour the species' movements. Moreover, as a result of forest management practices, small forest units may represent more intensively managed forest, which in turn could be unfavourable to the species' movements. All these aspects may be responsible for the fact that the *Rosalia longicorn* was more often found to occupy timber stacks situated within large, relatively intact, forest habitat patches.

Owing to the high density of timber stacks in the study area, the forest area free from their influence cannot act as an effective species refuge. Presumably, the spatial network of ecological traps leads to a functional fragmentation of the forest, free from their impact (Bruggeman et al. 2009;

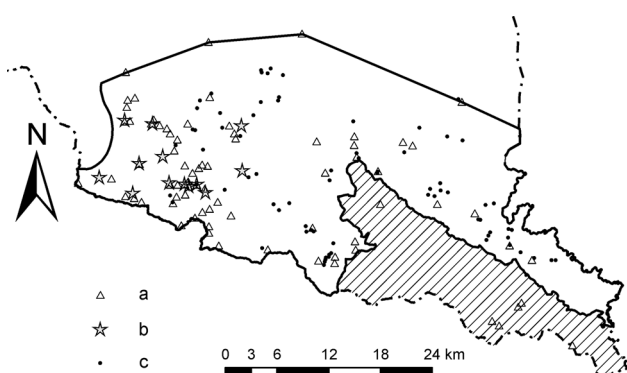


Fig. 5 Distribution of timber stacks as well as known localities of the *Rosalia longicorn* in the study area, (a) localities where the *Rosalia longicorn* was recorded between 2000 and 2013, (b) timber stacks surveyed in 2014–2016 where *Rosalia alpina* was present, (c) timber stacks surveyed in 2014–2016 where *Rosalia alpina* was absent

Lu et al. 2012), but this problem has not yet been studied or documented in depth. It is worth mentioning that the consequences of habitat fragmentation are not always limited to total habitat loss. In the least-cost patch approach, the increase in migratory cost is treated as an increase in isolation between subpopulations or habitat patches (Cushman et al. 2009). We recommend, therefore, that when the spatial structure of *Rosalia longicorn* habitat is studied, the distance from the nearest ecological trap should be included in analyses based on the resistant kernel approach (Compton et al. 2007). However, the insufficient amount of field data on *Rosalia longicorn* dispersal behaviour, mentioned earlier, is the real problem in reliable modelling of this process. Theoretically, most appropriate diffusion models of dispersal demand precise information on dispersal distances as well as the isolation power of different habitat types (Compton et al. 2007; Ovaskainen 2004, 2008). Such data with sufficient accuracy are available only for a few flying beetle species studied with radio-tracking techniques (Dubois and Vignon 2008; Tini et al. 2018). Unfortunately, none of those results can be used for estimating modelling parameters for the *Rosalia longicorn* in our study area.

The problem of the impact-free, functional fragmentation of a forest recalls the significance of large protected areas as effective species refugia (Hanski and Simberloff 1997; Ellis et al. 2011). In relation to the beech timber stacks in the Bieszczady Mts., only the Bieszczady National Park can be regarded as stack-free: it is thus a fairly large and intact habitat patch. This highlights the importance of the conservation method according to which a large protected area limits certain threats to a population.

The similarity between the mechanisms of the impact of timber stacks on xylophagous insects and that of light pollution should lead to similar conservation recommendations. Unfortunately, there are no data based on which the intensity of attraction of *Rosalia longicorn*s to stacked beech timber can be compared with suitable natural oviposition habitats. However, it is worth recalling that timber stacks are usually set up near forest roads on open, strongly insolated sites. Such placements increase the attractiveness of the stacks to the thermophilous *Rosalia longicorn* (Drag et al. 2011; Russo et al. 2011). The strong attraction of timber stacks for this species in the Bieszczady Mts. has been borne out by numerous field observations (Gutowski 2004b; Michalcewicz and Ciach 2012; Adamski et al. 2013, 2016). It should be stressed that these results only outline the potential area impacted by timber stacks in an ecological trap network; they do not provide any estimate of their influence on population condition.

The results of this study draw attention to one, hitherto not considered conservation topic: the possibility of reducing the area impacted by timber stacks by aggregating these: one large stack would affect a much smaller area

that several smaller ones. However, the lack of sufficient data on the luring mechanisms of timber stacks highlights the need for further investigation of this problem. This may also be useful from the point of view of conservation practice. As the dispersal potential of the *Rosalia longicorn* is still insufficiently understood, beech timber stacks in the species' range have to be monitored anyway. This activity would consume far less effort for a few large timber stacks than for numerous smaller ones. The above idea is consistent with the current conservation recommendation stressing the need to avoid storage of beech timber at sites and times accessible to the *Rosalia longicorn* (Gutowski 2004b; Michalcewicz and Ciach 2012; Adamski et al. 2016). However, recommendations to date have focused mainly on the time limitation, whereas the spatial approach has not been considered.

Conclusion

The results indicate that the *Rosalia longicorn* is exposed to strong pressure from timber stacks in managed forest areas during its mating and oviposition periods. The spatial distribution of timber stacks means that the majority of the population is endangered by ecological traps. The age and homogeneity of the forest stands surrounding timber stacks may contribute to the effectiveness of ecological traps. The same mechanisms may also be in operation in relation to other saproxylic species that are regularly observed wherever timber is stored (Müller et al. 2007). For conservation practice, it is crucial to avoid stacking beech timber during the *Rosalia longicorn*'s oviposition season; moreover, the concentration of numerous small stacks into a very limited number of bigger ones could reduce the spatial range of their impact.

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Compliance with ethical standards

Conflict of interest There are no conflicts of interest between the authors, the institution and fund donors.

Ethical approval The paper has been prepared and submitted in accordance with the ethical standards of the Committee on Publication Ethics (COPE). Moreover, the methodology used in this study involved only a low degree of invasiveness, so no ethical permit was required.

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