



## Comparison of tree microhabitat abundance and diversity in the edges and interior of small temperate woodlands



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### ABSTRACT

Forest edges are important features of wooded farmland. Their role for biodiversity was investigated using tree microhabitats (TMH) as an indirect indicator of forest biodiversity. Because they are managed more intensively, trees in the edges of fragmented temperate woodlands are likely to host more TMH than trees in their interior. In this study, we tested this hypothesis in relation to tree density, diameter, species composition and the structure and management of woodland edges. We selected 28 woodlands with edges differing by their structure and the adjacent fields. Eleven types of TMH were recorded in two transects set up in the edges and interior of the woodlands. TMH density was significantly higher in the woodland edges ( $4.67 \pm 0.78$  per 100 m<sup>2</sup> area) than in the interior ( $1.86 \pm 0.23$  per 100 m<sup>2</sup> area). Some TMH - patches of bark loss, cracks, sap runs and epiphytes - were significantly more abundant in the edges than in the interior. These results were accounted for by the convergence of several factors: (i) significantly higher tree density in woodland edges, (ii) a significantly higher proportion of TMH host trees in woodland edges, (iii) a larger tree diameter on average in woodland edges and, even in the same given size class, a higher frequency of TMH host trees, and (iv) greater abundance in woodland edges of tree species more likely to host TMH, even with small diameters. The positive relationship found between the height of the bottom of the crown and TMH abundance may have resulted from abiotic factors (micro-climatic conditions) related to long management rotations, but we did not demonstrate any direct effect of management practices. Given the large number of forest taxa, but also farmland taxa, that depend on TMH, woodland edges should be reconsidered as zones of high potential interest for forest biodiversity conservation.

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### 1. Introduction

Most of Europe's forest cover is made up of fragmented woodlands (Larsson, 2001), mostly in close association with farmlands. Their fragmentation is the result of habitat destruction (Tilman et al., 2001), but also of successive episodes of woodland clearance, cultivation and reforestation over historical time (Dupouey et al., 2002; Pitte, 2003). Fragmented forests are characterized by: (i) their small area, (ii) their isolation from other forest fragments and, as a result of the above, and (iii) a relatively higher proportion of woodland edges compared to the total land area at any scale in small woodlands as compared to large forests.

Forest edges are defined as an abrupt transition between relatively homogeneous ecosystems, at least one of which is forest

(Matlack and Litvaitis, 1999). Depending on the type of resources found in the two contiguous habitats (complementary or supplementary), different biodiversity responses are to be expected: neutral, positive or transitional (Ries et al., 2004). Empirical studies on the effects of forest edges on biodiversity have shown positive as well as negative effects, depending on which side of the edge is concerned, but biodiversity in forest edges is often higher than in the interior and some authors have identified specialist forest edge species (Ries and Sisk, 2010). Regarding forest biodiversity, the edge effect equates to a reduction of the core forest area, which can have a direct effect on the loss of forest species habitats (Hanski, 2005), but also an indirect effect by exposing organisms to the conditions of the surrounding ecosystems (Murcia, 1995). The study by Salek et al. (2013) is one of the few suggesting that forest edges could benefit forest biodiversity as some of their characteristics are favorable to species conservation, such as the presence of large trees and a corresponding potential for the

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occurrence of snags and high shrubs. They argue, moreover, that exploiting forest edges for timber production is of little economic interest, and conclude that woodland edges, in temperate riparian forests, should be dedicated to conservation. In farmland, however, woodland edges are managed by farmers and/or woodland owners to prevent forest growth from interfering with crop production or livestock breeding. The main management practices are the use of herbicides that could be detrimental to plant biodiversity in field margins (Le Coeur et al., 1997), or tree and/or branch cutting that could also promote biodiversity by creating tree microhabitats such as sap runs and patches of bark loss.

Tree microhabitats (TMH) are defined as structural heterogeneities on the trunks or branches of trees, such as cavities or cracks (Stokland et al., 2012). They host a wide range of taxa (Speight, 1989; Winter and Möller, 2008; Michel and Winter, 2009; Larrieu et al., 2012). Some taxa are highly dependent on TMH: more than 40% of birds in French forests use tree cavities or cracks for breeding (Blondel, 2005). Nearly 25% of the species hosted by forests depend on TMH or decayed wood for at least a part of their life cycle (Stokland et al., 2012). TMH are cited as relevant surrogates for taxonomic diversity assessment (Winter and Möller, 2008), at least for saproxylic beetles (Bouget et al., 2013, 2014). Furthermore, species that depend on saproxylic TMH (i.e. which are at least partially made up of decaying wood) contribute to nutrient cycling and thus to forest soil fertility (Speight, 1989). Studies on TMH abundance in large forests are related not only to tree species and tree diameter, but also to management (Vuidot et al., 2011; Larrieu et al., 2012; Regnery et al., 2013), but no studies have been conducted in small forests and woodland edges as considered here.

In comparing TMH in the interior and edges of forests, we tested the hypothesis that: (i) woodland edges support a higher TMH density than their interior, and (ii) some TMH will be more abundant in woodland edges. We explored the factors potentially related to TMH density in woodland edges: tree densities and diameters, tree species composition, edge structure and management practices.

## 2. Methods

### 2.1. Study area

The study was carried out in the *Vallées et Coteaux de Gascogne*, a long-term socio-ecological research site in a rural region in Gascony, south-western France (43°16N, 0°54E). The temperate agro-forested landscape is characterized by small woodlands embedded in a mosaic of meadows (livestock farming) and crops. Woodlands cover approximately 15% of the total area and most cover less than 20 ha. The region is hilly (250–400 m a.s.l.) and has a sub-Atlantic climate with slight Mediterranean influences (mean annual temperature 12.5 °C; mean annual precipitation 750 mm). These climatic influences are reflected in the medio-European type flora, where oaks (*Quercus robur* L., *Quercus pubescens* Willd., *Quercus petraea* Liebl.) are the main tree species, often in combination with hornbeam (*Carpinus betulus* L.), wild cherry (*Prunus avium* L.), wild service tree (*Sorbus torminalis* L.), sweet chestnut (*Castanea sativa* Mill.) and field maple (*Acer campestre* L.) (Gonzalez et al., 2009).

Previous studies in the region found no permanent spatial management units in the small woodlands, where logging areas were defined according to locally prevailing conditions (Du Bus de Warnaffe et al., 2006; Andrieu et al., 2011). Spatial logging patterns in these woodlands are therefore complex and hard to predict in terms of classic forestry rules. The woodland stands are characterized by coppice with standards growing from seed (ca. 30 ha). The coppices are generally assumed to be cut every 30 years by clear felling of small sections with farm machinery, mainly to produce

firewood. Some of the standards may be felled as well, for sale or own use as construction timber. Hedgerow trees are cut more frequently because of their faster growth rate and accessibility, for firewood and to manage field borders (Du Bus de Warnaffe et al., 2006).

Twenty-eight woodlands were selected in this study area, varying in size from 0.87 ha to 47.44 ha (mean  $\pm$  SE: 12.24  $\pm$  13.54), with the exception of one large 500 ha wood. All had at least one edge meeting the following criteria: (1) edges contiguous to agricultural land (meadows, oilseed rape or winter wheat), (2) straight edges at least 100 m in length; (3) separated from each other by at least 300 m; (4) no track, bank or ditch between the wood and the adjacent field. Because the orientation of woodland edges has been shown to induce different climatic conditions (Chen et al., 1993), with consequences for their biodiversity (Murcia, 1995; Ries et al., 2004), north-facing edges (“cold edges”) were excluded. Both sharp and smooth edges were selected according to their width and general physiognomy, with adjacent meadows or crop fields.

### 2.2. Observation of tree microhabitats

In each of the woodlands studied, we analyzed two parallel transects 100 m in length. The first transect was set up in the woodland edge. Its outer limit was the boundary of the wood, that is to say the line joining the first trees in the wood with a diameter at breast height (DBH) of at least 5 cm, and extending for 2.5 m inside the wood. However, trees further than 2.5 m from the boundary but with branches reaching the boundary, and therefore subject to field edge management, were also included in this “edge transect”.

The second transect was set up in the interior of the wood 50 m away from the first transect, but 5 m in width to compensate for the lower density of trees expected compared to the woodland edge, so that a similar number of trees could be observed in both transects. The distance of 50 m was chosen because the woods were at least 100 m wide at their narrowest point.

All the standing trees having their stem in the transects thus defined and a minimum DBH of 5 cm were then examined for the presence of microhabitats on the visible part of the trunk, both beneath and within the tree crown. Snags were taken into account and examined for their microhabitats only when they were more than 1 m in height. The DBH of all these trees, alive or dead, was also recorded and approximated to the nearest centimeter. The tree microhabitats observed are listed in Table 1: all were noted as the number of occurrences per tree, except epiphytes, which were recorded in percentage classes, and carpophores as present or absent.

We recorded a set of 11 microhabitat types as follows (see Table 1 for a full description of the main associated taxa): (i) **cavities excavated by woodpeckers**, with an entrance wider than 3 cm: included in this category were all woodpecker breeding holes and holes made by woodpeckers when feeding and deep enough to host a vertebrate; (ii) **cavities with mould** and an entrance wider than 10 cm: included in this category were other natural cavities and cavities with mould with an entrance more than 10 cm in width, as well as patches of missing bark of more than 100 cm<sup>2</sup> and wood at a decay stage of more than 3 (with reference to a 5-stage scale, see Table 1 SM), here, we differentiated cavities in contact with the ground (“base cavity”) or not (“trunk cavity”); (iii) **bark-lined cavities** (deep cavities formed between roots) with an entrance more than 3 cm in width; (iv) **cracks in the tree trunk** wider than 1 cm and more than 1 m above the ground, or bark in the process of peeling and forming a shelter. Dendrothelms were recorded when the entrance was wider than 3 cm, differentiating between (v) **rotted-bottom dendrothelms**, when the bottom of the cavity was made of decayed wood, and

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