Science of the Total Environment 619–620 (2018) 1319–1329

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Promoting biodiversity values of small forest patches in agricultural landscapes: Ecological drivers and social demand

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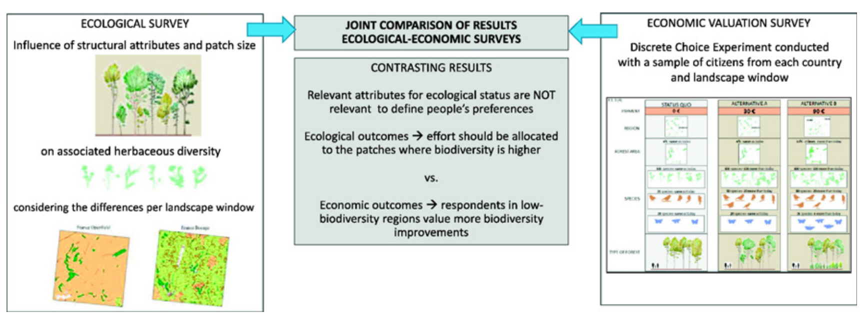
HIGHLIGHTS

- Ecologic and economic assessments conducted concomitantly, allowed a joint comparison.
- Patch size and tree species have the larger effect on plant species richness.
- Some key variables to improve biodiversity are not relevant to shape preferences.
- Some options preferred by people to increase biodiversity may be difficult to attain.
- Local population favoured policies improving biodiversity close to where they live.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 28 June 2017
Received in revised form 15 November 2017
Accepted 16 November 2017
Available online 23 November 2017

Editor: Simon Pollard

Keywords:
Economic valuation
Discrete choice experiment

ABSTRACT

Small forest patches embedded in agricultural (and peri-urban) landscapes in Western Europe play a key role for biodiversity conservation with a recognized capacity of delivering a wide suite of ecosystem services. Measures aimed to preserve these patches should be both socially desirable and ecologically effective. This study presents a joint ecologic and economic assessment conducted on small forest patches in Flanders (Belgium) and Picardie (N France). In each study region, two contrasted types of agricultural landscapes were selected. Open field (OF) and Bocage (B) landscapes are distinguished by the intensity of their usage and higher connectivity in the B landscapes. The social demand for enhancing biodiversity and forest structure diversity as well as for increasing the forest area at the expenses of agricultural land is estimated through an economic valuation survey. These results are compared with the outcomes of an ecological survey where the influence of structural features of the forest patches on the associated herbaceous diversity is assessed. The ecological and economic surveys show

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https://doi.org/10.1016/j.scitotenv.2017.11.190

Mixed models
Social preferences
Herbaceous diversity

contrasting results; increasing tree species richness is ecologically more important for herbaceous diversity in the patch, but both tree species richness and herbaceous diversity obtain insignificant willingness to pay estimates. Furthermore, although respondents prefer the proposed changes to take place in the region where they live, we find out that social preferences and ecological effectiveness do differ between landscapes that represent different intensities of land use. Dwellers where the landscape is perceived as more "degraded" attach more value to diversity enhancement, suggesting a prioritization of initiatives in these area. In contrast, the ecological analyses show that prioritizing the protection and enhancement of the relatively better-off areas is more ecologically effective. Our study calls for a balance between ecological effectiveness and welfare benefits, suggesting that cost effectiveness studies should consider these approaches jointly.

1. Introduction

In Europe, the conversion of forests into agricultural land and the intensification and specialization of agriculture since the 1950s has led to reduction and fragmentation of the original forest cover, to decreased landscape heterogeneity and ultimately, to a decline of species diversity (Foley et al., 2005; Hadad et al., 2015; Valdés et al., 2015).

Small forest patches embedded in agricultural (and peri-urban) landscape matrices in Western Europe are often overlooked in conservation programmes, although they play a key role for biodiversity conservation as they often are the only semi-natural habitats present in these landscapes. Furthermore, their capacity to deliver a whole suite of ecosystem services (ES) to society (e.g. recreation opportunities, food production, pest control) is increasingly recognized (Decocq et al., 2016; Foley et al., 2005; Valdés et al., 2015). Due to their small size, these patches are generally not legally protected against conversion to another land use or against any other form of degradation. Hence the need for policies that can maintain and restore biodiversity in these small forest patches.

Many of the benefits that biodiversity conservation policies provides are public goods not traded in markets. Hence, considering only financial costs and benefits of these policies may produce sub-optimal decisions in terms of their ability to optimize social welfare. Environmental valuation can help guiding the design of these policies by eliciting public preferences on different attributes of biodiversity (Fatemeh Bakhtiari et al., 2014; Christie et al., 2006), so these can be taken into consideration in investments and policy decisions (Stenger et al., 2009). Proposed measures should be both socially desirable and ecologically effective. This includes considerations on where - under which landscape conditions, changes will be valued the highest, will have largest effect on biodiversity changes, and will be most expensive. Hence there is a need for integrated ecological - economic research in which the factors determining biodiversity patterns in these patches are identified together with the preferences of the local population for improved biodiversity and management measures leading to a better conservation status.

We hypothesize that social support may exist for preserving and enhancing the status of these small forest patches. However, social preferences may vary depending on the management measures undertaken and the type of landscape where these are applied (van Zanten et al., 2016). Also, we hypothesize that less public support and lower ecological effectiveness can be expected for biodiversity oriented measures in landscapes that provide more habitat and suffered less degradation (Domínguez-Torreiro et al., 2013; Horowitz et al., 2007).

Based on these hypotheses, this study has three main objectives:

1. to analyse the social preferences for biodiversity-oriented measures in small forest patches in agricultural landscapes, using both species and structural diversity indicators;
2. to analyse the ecological effectiveness of the proposed measures in these landscapes;
3. to determine whether the social preferences and effectiveness differ between landscapes with different degrees of agricultural management intensity.

To address these objectives, a joint ecological and economic assessment was conducted on small forest patches in Belgium (Flanders) and northern France (Picardie). In each study region, two contrasting types of agricultural landscapes were selected: open field (OF) and bocage (B). These landscape types result from different historical trajectories and show different biodiversity conservation levels; OF landscapes are characterized by large-scale, high input-based agriculture while in B landscapes a more small-scale, lower-input agriculture is practised. The connectivity between the forest patches in the B landscapes is considered to be higher due to the high number of treelines and hedgerows compared to the OF landscapes.

The social demand for enhancing key biodiversity components, forest structural components as well as for increasing the forest area at the expenses of agricultural land is estimated through an economic valuation survey. Results are compared with the outcomes of an ecological survey where the biodiversity levels in OF and B landscapes are assessed, together with the influence of structural features of these stands on the associated herbaceous diversity. This indicator is adopted due to its impact on multi-trophic interactions that seem to indicate its suitability as biodiversity indicator (Scherber et al. 2010).

This work contributes to the still limited number of studies addressing the role that forest patches in agricultural landscapes play in the conservation of biodiversity and in the provision of ES (Mitchell et al., 2014; Valdés et al., 2015), being one of the main novelties that ecologic and welfare economic assessments were conducted concomitantly, thus allowing a joint comparison of the key attributes that play a decisive role in determining biodiversity patterns, and their contribution to shape social preferences for these forest patches.

2. Methods

2.1. Study area

Both in Flanders and Picardie, two 5 × 5 km landscape windows (LW) with contrasting agricultural management intensities were selected (Figs. 1 and 2). One window in each region (hereafter 'Open Field Landscape', OF) was composed of isolated forest patches embedded in an intensively cultivated agricultural matrix dominated by arable land, with big crop fields (from one to several hectares) receiving high inputs of chemical fertilizers and biocides annually. The other window (hereafter 'Bocage Landscape', B) contained forest patches that were more or less connected by hedgerows, embedded in a matrix dominated by grasslands and small crop fields (usually <1 ha) that were less intensively managed and received far less inputs.

The forest cover represented 5.4, 6.4, 4.7 and 5.4% in the Belgian B, Belgian OF, French B and French OF LW, respectively, distributed

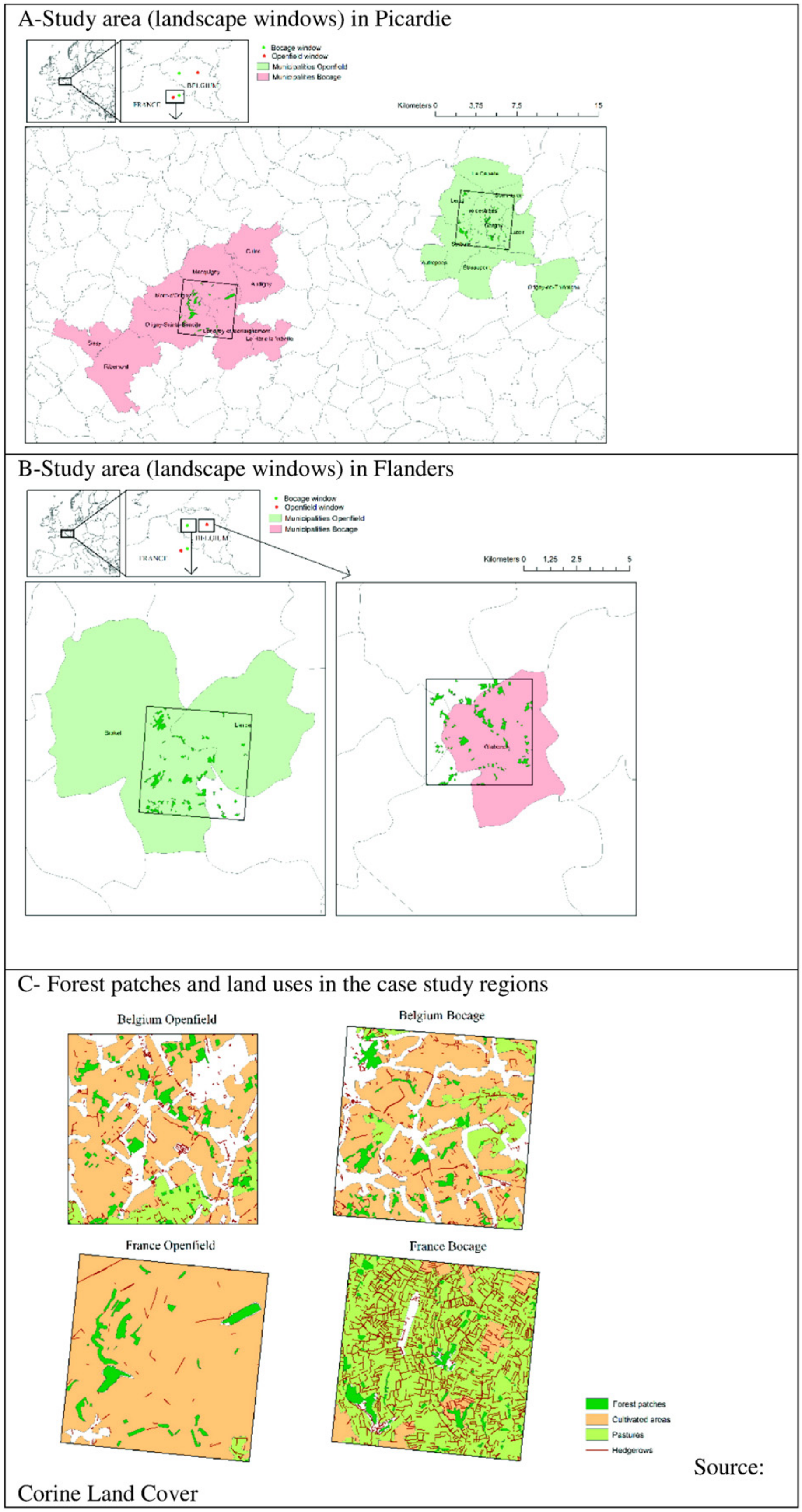


Fig. 1. Study area. A and B: Municipalities in the Openfield window in green. Municipalities in the Bocage window in pink. C. Forest patches are shown in dark green, pastures in light green, cultivated areas in light orange and hedgerows are shown as dark red lines.

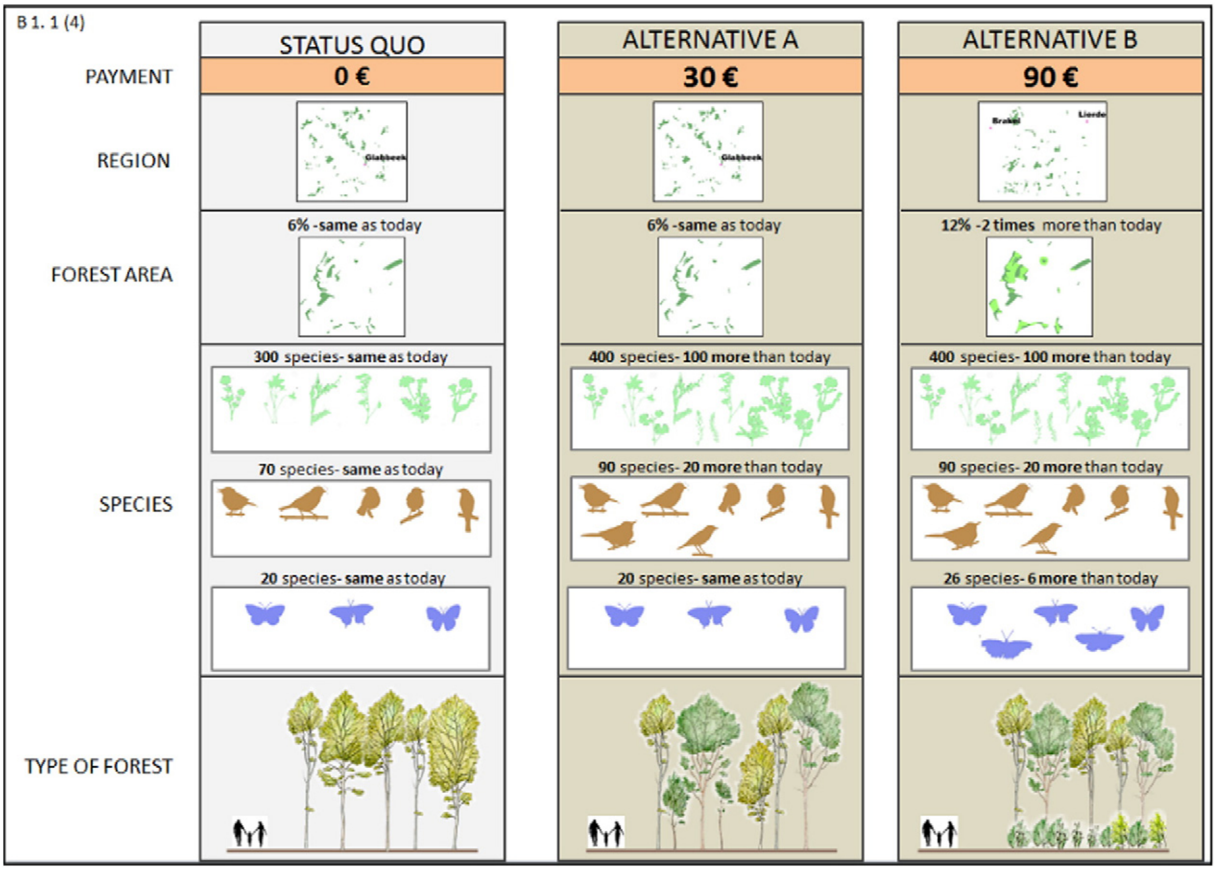


Fig. 2. Example of a choice card shown to the respondents in the Flanders region.

among 56 (min: 0.24 ha, mean: 2.43 ha, max: 22 ha), 67 (0.17, 2.40, 16), 62 (0.09, 1.89, 27), and 29 (0.17, 4.67, 24) patches, respectively.

The valuation survey was conducted in the municipalities located within and around the landscape windows (see Fig. 1).

2.2. Economic valuation

Discrete choice experiment (CE) is an attribute based method rooted on the Lancaster's theory of value (Lancaster, 1966; Train, 2009) and the random utility theory (McFadden, 1974). Lancaster theory (1966) states that the utility that an individual derives from a good consists of the sum of the value of all the attributes of that good. In random utility theory (McFadden, 1974), respondents try to maximize utility functions that consists of a deterministic and a stochastic element.

DCE involves the characterization of the good or service at stake, i.e. forests patches, through a series of its most relevant attributes that are combined to create hypothetical scenarios or alternatives that will be evaluated by the respondents, by choosing their preferred scenario. One of the attributes included is a monetary attribute enabling to calculate willingness to pay (WTP) estimates for each of the remaining attributes as well as for each of the given alternatives. The econometric specifications and details on the method are intensively written in the literature, and will therefore not be repeated here. We refer to Louviere et al. (2000), Haab and McConnell (2002) and Johnston et al. (2017) for specifications and applications.

A DCE was conducted on a representative sample of the local population for each LW. The DCE enables capturing both use values (recreational and aesthetic enjoyment) and non-use values (existence values) that people may associate to the biodiversity of these patches. A set of ecologically relevant attributes was defined (see Table 1) together with forest ecologists in the team and after a careful review of economic valuation literature on forest-related biodiversity.

Table 1
Attributes and levels used in the setup of the DCE. Biodiversity and forest structure attributes were continuously coded after testing effects coding with no satisfactory results. LW attribute was dummy coded (Open field - 0 and bocage - 1)

Attribute		Levels
Landscape window	LW	Open field
		Bocage
Forest area	Area	6%[a]
		9% (1.5 times more than today)
		12% (2 times more than today)
Biodiversity - Herbaceous species	Herb	300[a]
		350 (50 more than today)
		400 (100 more than today)
Biodiversity - butterfly species	Butter	20[a]
		23 (3 more than today)
		26 (6 more than today)
Biodiversity - bird species	Bird	70[a]
		80 (10 more than today)
		90 (20 more than today)
Forest structure - tree species	TSP	1[a]
		2
		3
Forest structure - shrub layer	Lay	Tree layer with NO shrub layer[a]
		Tree layer with shrub layer
Forest structure - tree ages	Ages	1 age[a]
		2 ages
Cost (€)	Cost	0[a]
		10
		30
		50
		70
		90
		110

[a] Attribute levels corresponding to the current scenario or status quo (SQ). For the landscape window attribute, we controlled for the respondents in each of the LW locations, so that they were provided the SQ alternative corresponding to the LW where they lived.

An attribute with two levels presented the LW where the management measures would take place: open field (OF) or bocage (B). This attribute allowed testing whether respondents were sensitive to the location of the proposed changes.

An attribute with three levels addressed the area covered by forest patches in these LW. The current level or status quo (SQ) level was set on 6% forest cover; two additional levels presented an increase up to 9% (1.5 times more than today) and 12% (2 times more than today) forest cover, respectively. Fragmentation of forest cover is a key issue for many species in these landscapes, leading to isolated populations for species having more limited dispersal capacity (Lindborg and Eriksson 2004, in Lindborg et al., 2009) and to an increased edge:core ratio detrimental to forest species. Accordingly, the increase in forest area was spelt out to the respondents as always taking place enlarging and connecting existing forest patches. The proposed area enlargement by forest patch connection would be in line with existing policies to tackle fragmentation of natural habitats (IEEP and Alterra, 2010), reducing the isolation of the forest patches, and enhancing their role as refugia for forest specialist species (Roy & de Blois, 2008; Araujo Calçada et al., 2013, in Valdés et al., 2015; Maguire et al. 2015, Mitchell et al., 2014).

A group of three attributes presented structural features of the forest patches key to improve biodiversity levels and dynamics of these ecosystems and have been previously addressed in valuation studies (e.g. Nielsen et al., 2017; Meyerhoff and Liebe, 2008; Campbell et al., 2014; Filyushkina et al., 2017). The attribute on tree species richness considered three levels, departing from one species and increasing up to three tree species. The age attribute considered one age (even-aged) or two age (uneven-aged) tree stands. The layer attribute considered the absence or presence of a shrub layer.

Three attributes considering herbaceous, butterfly and bird species covered the species dimension of biodiversity. Herbaceous species is the associated diversity indicator assessed in the ecological analysis (see below) as it constitutes greater part of temperate forest biodiversity (Gilliam, 2007)., Two other taxonomic groups were included to test whether preferences vary among different taxonomic groups (Home et al., 2009; Martín-López et al., 2007). Levels for these attributes were derived from secondary data on inventories in the study areas while expected increases were considered based on the size of the regional habitat species pool (i.e. the number of species potentially present in the study sites if habitat conditions become suitable). For French LW, we used the CLICNAT (http://obs.picardie-nature.org) and DIGITALE 2 (http://www.cbnbl.org/) databases for the fauna and flora, respectively. For the Belgian windows information was acquired from Van Landuyt et al. (2006) for plants, Vermeersch et al. (2004) for birds and Maes et al. (2013) for butterflies added with recent data from the online database waarnemingen.be (http://www.waarnemingen.be).

Finally, a monetary attribute for the estimation of willingness to pay (WTP) was included. Levels were based on a similar study recently conducted in Flanders (Liekens et al., 2013). The payment vehicle was a one-time mandatory payment per household and directly allocated to a fund ruled by the regional government and monitored by the local community council and by the University of Ghent and Picardie, respectively.

2.2.1. Questionnaire design and administration

A questionnaire was designed to implement the DCE (see Appendix A). The questionnaire was tested in pilot test with a total of 20 respondents prior final launching. Within each window the sample was stratified according to age and gender, proportional to the population of each window. Our sample had an overrepresentation of middle-age and elder age classes compared to the real population.

The SQ option depicted monospecific even-aged forest patches without a shrub layer, covering 6% of the landscape area and hosting the lowest number of herbaceous, bird and butterfly species respectively within the ranges considered. The SQ level for the landscape window was case-sensitive, so it would show for each of the subsamples their window where they belong to. The groupings of SQ and the proposed alternatives are known as choice sets. In this case, each choice set involves the SQ option and two alternatives. 24 choice sets were designed using a pivot experimental design optimized by NGene (ChoiceMetrics, 2012) for D-efficiency, retrieving a D-error of 0.0022. The valuation questionnaire consisted of an introductory section, a valuation section with six choice sets per respondent (see Fig. 2) and follow-up questions on socio-economic characteristics. Additionally, in the French survey space for respondents' comments was included.

A total of 449 valuation questionnaires were completed in face-to-face surveys, 242 in the Flemish LW and 207 in the French LW, between August 2013 and August 2014. The questionnaire was delivered to a sample of the population equally weighted across the OF and B areas in France and Belgium and sampled from municipalities closest to the forest patches (see Appendix A). Within each window the sample was stratified according to age and gender, proportional to the population of each window. Forty-eight (10.7%) protest answers were identified through a follow-up close-ended question. Protesters were mainly people stating that they already pay enough taxes and that the government should pay for these initiatives (cf. Meyerhoff et al., 2014). The share of protest answers is lower than this found in similar studies conducted in other European countries (Meyerhoff and Liebe, 2008; Meyerhoff et al., 2012; Varela et al., 2014; Valasiuk et al., 2017).

2.2.2. Econometric model

Random Parameter Logit (RPL) models are flexible estimation methods that are being increasingly employed to model people's preferences within the random utility framework (Train, 2009). All attribute parameters related to the forest patches were assumed to be random and to follow a normal distribution, thereby allowing assessment of heterogeneity in these parameters. The cost attribute parameter was assumed to be fixed as we wished to restrict it to be non-positive for all individuals (Train, 2009). A maximum likelihood estimation of the model parameters was conducted in NLOGIT 5.0 (Greene, 2007) using simulation with 500 Halton draws.

2.3. Ecological assessment

2.3.1. Data collection

In 2012, all forest patches in both windows were surveyed for all vascular plant species at the peak of plant phenology, including all herb, shrub and tree species. Herb species were subsequently split into two non-overlapping groups: «forest specialists», i.e. species belonging to forest phytosociological classes according to Oberdorfer et al. (1990), modified to include some species restricted to forests in our study area; and «generalists», i.e. species found in forests but having their optimum either in forest-associated habitats (e.g. edges, clearcuts) or in non-forest habitats (e.g. grasslands, crop fields). To comprehensively survey vegetation, we walked along parallel transects located 10-m apart from each other and recorded all vascular plant species. We thus obtained a value of species richness per patch for each herb group as well as for woody plants.

The drivers to explain variations in herbaceous plant species richness among patches were aligned with the survey attributes and included: patch area, patch age, tree species diversity, tree diameter coefficient of variation, density of the shrub layer. We used patch area and age as potential drivers of plant species richness: smaller forest patches might host less species (Jacquemyn et al., 2001) according to the species-area relationship (Rosenzweig, 1995; Paal et al., 2011); similarly, recent forest patches may host less species than mature ones according to the species-time relationship (Rosenzweig, 1995), especially with respect to forest specialists (Hermy and Verheyen, 2007; De Frenne et al., 2010). Forest patch area was calculated using a GIS and digitized aerial photographs, all taken after the year 2000. Patch age was estimated on the basis of the date of the oldest map on which a patch was represented for the first time, using old maps from the

Table 2
RPL results for the open field and bocage landscapes. Results correspond to taste parameters which measure the intensity of preferences (utility) that respondents have for the different attributes and their levels as shown to them in the choice sets. Mean coefficient distribution indicates the mean value for the attribute. Because a normal distribution was assumed for the non-monetary parameters, significant standard deviation of a parameter distributions indicates that the attribute is heterogeneous around the mean, i.e. not all the respondents have the same preferences for it.*

ATTRIBUTES	Respondents living in areas with open field landscape		Respondents living in areas with bocage landscape	
	Mean coefficient of distribution (s.e.)	s.d. of parameter distributions (s.e.)	Mean coefficient of distribution (s.e.)	s.d. of parameter distributions (s.e.)
LW (landscape window)	−1.459 (0.451)***	3.218 (0.487)***	2.652 (0.402)***	2.8160 (0.435)***
Area (% area covered by forests)	0.182 (0.07)***	0.5088 (0.1045)***	0.263 (0.061)***	0.443 (0.670)***
Herb (n° of herbaceous species)	0.010 (0.007)	0.047 (0.009)***	0.009 (0.005)*	0.036 (0.006)***
Butter (n° of butterfly species)	0.137 (0.061)**	0.4123 (0.099)***	0.033 (0.056)	0.362 (0.085)***
Bird (n° of bird species)	0.126 (0.038)***	0.217 (0.046)***	0.054 (0.022)**	0.152 (0.029)***
TSP (n° of tree species)	0.200 (0.191)	1.294 (0.386)***	0.268 (0.163)	1.218 (0.258)***
Lay (having a shrub layer)	1.790 (0.435)***	3.452 (0.614)***	0.627 (0.265)***	1.917 (0.390)***
Ages (n° of tree ages)	1.806 (0.219)***	3.130 (0.612)***	0.540 (0.277)*	2.150 (0.368)***
Cost (payment per household)	−0.048 (0.007)***	Fixed	−0.048 (0.006)***	Fixed
ASC (alternative-specific constant)	−1.8659 (0.5107)***	Fixed	−1.4696 (0.3951)***	Fixed
Pseudo - r2	0.3288		0.2769	
Log-likelihood function	−884.188		−949.305	

s.e.: standard error, s.d.: standard deviation ns (not significant).
* $p < 0.10$.
** $p < 0.05$.
*** $p < 0.01$.

18th, 19th and 20th centuries. As a given patch may contain a mosaic of fragments with different ages, we calculated an area-weighted average of the age of all fragments composing a patch.

Forest canopy and structural diversity are well-known drivers for many taxonomic groups (e.g. birds and butterflies (Tews et al., 2004) and also for vascular plants (Ampoorter et al., 2016). The canopy diversity variables were quantified in a subset of 16 forest patches in each LW. To guarantee representative selection of the variation of patch size and patch age into each window and for that purpose we divided the patches in two categories of size (small vs. large patches) and age (old vs. recent patches), distinguished by the respective median values of, respectively, size and age as division points between categories. Four patches for each of the four combinations of size x age categories (small-old, small-recent, large-old, large-recent) were selected, ending up with a subset of 16 patches per window.

Forest structure has been determined based on the PCQ-Method (Cottam and Curtis, 1956). Two trees per quarter within 20 m of a sampling point have been measured for height, diameter at breast height (d130) distance and angle to the theoretical central point and their species has been recorded. These two trees per quarter were distinguished from one another by being smaller or larger than 30 cm d130 to sample information about different age groups within the forest stand. The tree closest to the theoretical central point has additionally been utilised to determine the same characteristics of the "structural group of four" (Pommerening, 2002), a group of five trees usually in close vicinity to one another. Diameter values have been used to calculate the diameter coefficient of variation and the species identities to calculate true shannon diversity (Jost, 2006).

Density of shrubs is based on the availability of phanerophytes with stems <7 cm average diameter and a height of >1.3 m in a radius of 2 m around the sampling point.

2.3.2. Data analysis

Total herb and forest herb specialist richness per patch were used as response variables in linear mixed models with the region (Flanders vs Picardie) as a random factor. We used landscape type (B versus OF), patch size, patch age and the three canopy variables (tree species diversity, tree diameter variability and shrub cover) as fixed factors. The latter variables were only available for a subset of patches. Therefore, models including all patches and only landscape type, patch size and age as fixed factors were fitted as fixed factors. In models using the subset of 64 patches all fixed factors were included. To meet homoscedasticity requirements, the variables 'patch size' and 'shrub cover density' were ln-transformed prior to analyses. All analyses were performed with SPSS, version 23.

3. Results

3.1. Social preferences results

We focused on exploring heterogeneity in preferences between OF and B subsamples by pooling the two-country data together (see Table 2). We corrected for the scale parameter prior sample merging.

Table 2 shows the results of the preference parameters[3]. The sign of the LW attribute (0 for open field level and 1 for the bocage level) parameter indicates that respondents in both landscape types would prefer to have the proposed changes implemented in their own window. Also, both samples retrieved negative values for the alternative specific constant (ASC), indicating, ceteris paribus, a willingness to depart from the SQ scenario towards alternative scenarios. Similar preference patterns are encountered across the two subsamples with the tree species attribute not being significant in determining people preferences. Regarding the species set of attributes, bird species do retrieve significant and positive results in both cases; the herbaceous diversity has low or no significance (bocage and open field, respectively) in shaping people's preferences, similarly to the butterfly species (significant for open field subsample and no significant for bocage subsample).

Table 3 presents the Marginal Willingness to Pay (MWTP) estimates for each of the two subsamples. In general, we see that OF respondents show higher MWTP values than their B counterparts for increasing the number of species of different taxonomic groups or enhancing the forest structure, whereas respondents in the B region are more concerned about having these policies implemented in their own region and increasing the forest area while caring less about the resulting forest structure or species richness.

Table 4 presents six different policies relevant for the management of these small forest patches and the gains in welfare these would represent in each LW with respect to the SQ scenario. Policies from 1 to 4 represent changes in the attributes liable to be influenced by forest management and in one attribute at the time to better illustrate the gains in welfare. Promoting a shrub layer produces the highest gains in welfare in both windows. Increasing the number of tree species

[3] Due to perfect scale confounding effects, direct value comparison of preference parameters across subsamples cannot be undertaken, while WTP estimates are scale-free and hence directly comparable across subsamples.

Table 3
Results of the WTP estimates. Estimates of willingness to pay were calculated for both subsamples employing Delta method. Confidence intervals were estimated following the Krinsky and Robb method with 1000 draws (Krinsky and Robb, 1986). Continuous coding was employed for all the attributes (previous testing of effects and dummy coding did not result in significant results). LW was coded such that 0 correspond to open field and 1 to bocage. The rest of the attributes were continuously coded.*

	Open field subsample		Bocage subsample	
Attributes	WTP per unit of the attribute (s.e.)	95% Confidence interval	WTP per unit of the attribute (s.e.)	95% Confidence interval
LW	−30.66 (9.589)***	(−49.45, −11.86)	55.27 (8.17)***	(39.26, 71.27)
Area	3.82 (1.458) ***	(0.96, 6.68)	5.48(1.18)***	(3.17, 7.80)
Herb	0.20 (0.143)	(−0.08, 0.48)	0.19 (0.12)	(−0.04, 0.42)
Butter	2.88 (1.26465)**	(0.40, 5.36)	0.69 (1.13)	(−1.53, 2.91)
Bird	2.66 (0.765)***	(1.16, 4.16)	1.13(0.51)**	(0.12, 2.13)
TSP	4.20 (4.177)	(−3.98, 12.39)	5.58 (3.55)	(−1.38, 12.55)
Lay	37.61 (9.420)***	(19.15, 56.08)	13.10 (5.68)**	(1.97, 24.22)
Ages	24.07 (8.257)***	(7.88, 40.25)	11.26 (5.65)**	(0.19, 22.32)

s.e.: standard error.
* $p < 0.10$.
** $p < 0.05$.
*** $p < 0.01$.

does not produce any change in the welfare of either regions. OF respondents are less sensitive to policies increasing the forest area, whereas a structural change such as increasing tree ages retrieves similar welfare gains. The remainder two policies (5 and 6) respectively show how a hypothetical maximization of the number of species and a hypothetical maximal improvement on the structural diversity would impact the welfare in each of the windows. Open field respondents would benefit more from an optimal increase in species while wellbeing of bocage respondents would be higher in a maximal structure diversity scenario.

3.2. Ecological results

The outcomes of the mixed models (Table 5) indicated that both total and forest herb specialist richness strongly increased with patch area. Herb species richness was also significantly higher in the B landscapes: on average 12 to 16 more herb species and 5 to 7 forest herb specialists occur in the B landscape patches relative to the OF landscape patches (Fig. 3). Patch age only significantly affected forest herb specialist richness when all patches were included, although a similar trend was observed in the reduced dataset. Among the canopy variables, only tree species diversity had a (consistently) positive impact on herb species richness.

4. Discussion

This study provides insights into the ecology and the social preferences for the main features of small forest patches in agricultural landscapes in Western Europe. Results show that people prefer biodiversity improvement measures to take place close to where they live, but the type of improvements preferred differ across landscape windows. We hypothesize that these differences may be related to the functional interpretation people have of biodiversity and potentially also to the opportunity cost that changes in the land use may have. Comparison of ecological and economic analysis reveals that some of the options preferred by people to increase biodiversity may prove difficult to attain; also, some of the key variables to improve biodiversity levels are not relevant to shape people's preferences.

4.1. The economic valuation of biodiversity-related attributes

The results show that social support exists for preserving and enhancing the status of the forest patches; and that these preferences are location-sensitive, i.e. respondents favoured policies that improved biodiversity close to where they live. This is in line with findings from other studies (e.g. Dallimer et al., 2015).

Interestingly, OF interviewees show more interest in improving the biodiversity content of those patches that already exist. This may be

Table 4
Compensating surplus (CS) estimates for the different policies and for both landscape windows. Each policy represents a change from the SQ level for the attribute in bold. The compensating surplus estimates show the gains in welfare, in terms of € per household that average dwellers would experience as a result of the implementation of that policy.

Policy 1	Increase the forest area								CS estimates (€/household)	
	LW	Area	TSP	Lay	Ages	Herb	Butter	Bird	Open field	Bocage
Levels	0/1	**9**	1	0	1	300	20	70	11.46	16.44
Policy 2	Increase the number of tree species								CS estimates (€/household)	
	LW	Area	TSP	Lay	Ages	Herb	Butter	Bird	Open field	Bocage
Levels	0/1	6	**3**	0	1	300	20	70	0.00	0.00
Policy 3	Increase the number of tree ages								CS estimates (€/household)	
	LW	Area	TSP	Lay	Ages	Herb	Butter	Bird	Open field	Bocage
Levels	0/1	6	1	0	**2**	300	20	70	24.07	11.25
Policy 4	Promote the existence of a shrub layer								CS estimates (€/household)	
	LW	Area	TSP	Lay	Ages	Herb	Butter	Bird	Open field	Bocage
Levels	0/1	6	1	**1**	1	300	20	70	37.61	13.1
Policy 5	Maximize number of species								CS estimates (€/household)	
	LW	Area	TSP	Lay	Ages	Herb	Butter	Bird	Open field	Bocage
Levels	0/1	6	1	0	1	**400**	**26**	**90**	70.28	22.4
Policy 6	Maximize structure diversity								CS estimates (€/household)	
	LW	Area	TSP	Lay	Ages	Herb	Butter	Bird	Open field	Bocage
Levels	0/1	6	**3**	**1**	**2**	300	20	70	61.68	24.35

Table 5
Parameter estimates of the linear mixed models fitted to explain total and forest herb species richness in forest patches located in the B versus OF landscape windows in Flanders and Picardie. The results of both the analysis including all patches (left columns) and the subset of 16 patches per window (right columns) are presented.

Variable	Total herb species richness		Forest herb species richness	
	All patches	Subset of 16 patches/window	All patches	Subset of 16 patches/window
Patch area $	13.80***	14.45***	4.69***	5.12***
Patch age	−0.02 ns	0.01 ns	0.02***	0.02 ns
Landscape type £	12.61***	16.75***	4.70***	6.55**
Tree species diversity		3.41*		1.57**
Tree diameter variability		−9.55 ns		−2.93 ns
Density of shrub layer $		1.44 ns		0.40 ns

$: ln-transformed; £: O is reference; ns.
* p < 0.10.
** p < 0.05.
*** p < 0.01.

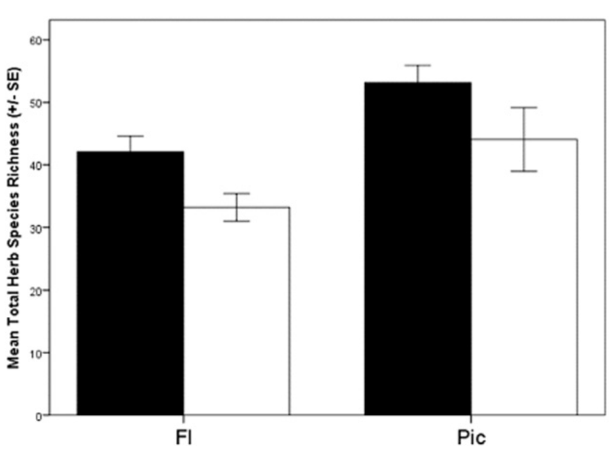
due to diminishing marginal utility of biodiversity - as economic theory would also predict. However, it provides an interesting result from a welfare economic perspective since efforts should then be allocated to areas with low biodiversity today - assuming that the biodiversity increase obtained per effort is the same. This is quite likely not so, but would require cost estimates to be considered, see e.g. Nielsen et al. (2017) who consider this aspect (but not the assessment of social preferences).

Most valuation studies addressing biodiversity through choice experiments use the number of species as the attribute to convey biodiversity (e.g. Horne et al. 2005, Hoyos et al. 2012, Juutinen et al. 2011) as it is regarded by the public as one of the most frequent characteristics when conceiving biodiversity (Bakhtiari et al., 2014). While using generic species may be taken as an indicator of biodiversity (Varela et al., 2014), conveying biodiversity through the status of either iconic species (e.g. Loomis and González-Cabán, 1998), generic endangered species (e.g. Campbell et al., 2014; Tyrväinen et al. 2014) or specifically named endangered species (Jacobsen et al., 2008) may lead to very high, potentially overestimated, values of species preservation (Jacobsen et al., 2008). Our study contributes to this literature by showing that even if we use the number of species as an attribute, the value people attach to it may differ depending on which group the species belong to. Birds being the most preferred, followed by butterflies, and herbs and tree diversity valued much less. Our results are in line with research showing that use values (in this case linked to birdwatching and knowledge of most common bird species), together with phylogenetically closeness to humans may have played an important role in determining preferences (Martín-López et al., 2007, 2011; Morse-Jones et al., 2012. While we may speculate on the reasons for the results, the implication is that even the use of number of species as a measure in valuation may need to be refined for evaluation to more specific groups.

In our study the tree species attribute retrieved no significant WTP estimates in either region. This is in contrast with previous studies (e.g. Filyushkina et al., 2017, Varela et al., 2014). One potential reason is that the recreational dimension of the small forest patches is limited by their size, and so the aesthetic experience may have a more relevant role than in standard forest-people interaction (Decocq et al., 2016). In this sense, the fact that in our study the proposed changes take place in deciduous stands (i.e. no change from coniferous to mixed or deciduous stands) has a lower impact on the aesthetic features of the forest patches compared for example to changes from coniferous to mixed or to deciduous stands.

The inclusion of structural features of these stands beyond generic number of species is aligned with studies where biodiversity is not only addressed as richness in species but also considering the role it plays as a regulatory of ecosystem processes and functions (Bakhtiari et al., 2014). Studies such as these conducted by Christie et al. (2006), Czajkowski et al. (2009), Eggert and Olsson (2009), McVittie and Moran (2010), Campbell et al. (2014) and Bakhtiari et al. (2014) consider both functionality (e.g. opportunity for natural processes in the forest (Campbell et al., 2014)) and value of biodiversity as a good in itself.

This set of structural attributes can be considered by some respondents as final attributes or outcomes of a given management policy, i.e. obtaining a change in the forest structure that enhances their recreational or aesthetic experience. However, these can also be regarded as intermediate or causal attributes, i.e. changes in the forest structure may increase diversity in a series of taxonomic species. Johnston et al. (2017) signal that including causal attributes and failing to include final outcome attributes in valuation surveys may bias welfare estimates, as the valuation scenario leaves open the possibility for the respondents to speculate for the omitted outcomes. Hence, the inclusion of a variety



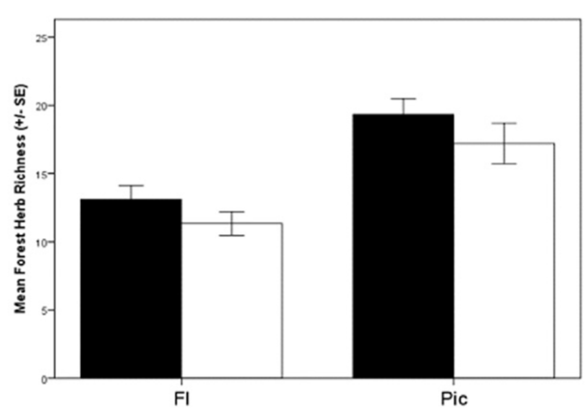


Fig. 3. Mean (± standard error) total (a) and forest herb species richness (b) across all forest patches in the forest patches in the B (black bars) and OF (white bars) landscape windows in Flanders (Fl) and Picardie (Pic).

of taxonomic diversity and structural attributes would prevent against this bias.

To illustrate this discussion, policies 5 and 6 in Table 4 show the result of maximizing either outcome or causal attributes, respectively. Indeed these policies overlook ecological rationality (since changes in structural and taxonomic species are interwoven), but illustrate the different preferences in each region, with higher welfare gains for open field respondents when policies optimize the delivery of taxonomic species while bocage respondents obtain higher welfare estimates when the structural diversity of the forest is maximized.

4.2. Outcomes of the ecological analyses

The outcomes of the ecological analyses are in line with the expectations. Larger patches hosted more species as predicted by the species-area relationship (Rosenzweig, 1995). Also the species-time relationships are in line with earlier findings (e.g. Jacquemyn et al., 2001). Forest herb species richness increased with patch age, which can be attributed to the often limited colonization capacity of many forest herbs (e.g. De Frenne et al., 2010). The rather limited strength of this forest species-time relationship is likely due to the fact that in our analyses a given patch may contain a mosaic of fragments with different ages, which adds noise to the species-time relationships. The absence of a relationship between patch age and total species richness has been found before (e.g. Jamoneau et al., 2011) and it is likely explained by a gradual substitution of non-forest herbs, often associated with the land use prior to afforestation, with forest herbs as the forests become older. The effects of the land-use intensity surrounding the patches was very consistent, with a clearly lower total and forest herb species richness in patches located in the more intensively managed OF landscapes. These patterns are in accordance with models predicting the effects of the surrounding landscape matrix on local species richness (cf. Tscharntke et al., 2005) and with the results of Jamoneau et al. (2011) in a similar context. Finally, we observed that tree species diversity was the forest canopy variable that most strongly affected the (forest) herb species richness. Our results confirm the forest level findings by Ampoorter et al. (2016). Although data is lacking to identify the exact mechanism, we suspect that the positive effect of tree species diversity is in this case most likely caused by the different environmental conditions created by combining multiple tree species in a single patch. The other forest canopy variables appeared less important for herb species richness, but it is not unlikely that they will impact the diversity of birds and butterflies (Tews et al., 2004), the other taxonomic groups that figured in the questionnaire.

Summarizing the ecological data analysis clearly pointed out that larger, older patches with a diverse tree layer and located in the B landscapes are most rich in (plant) species. Conservation of these patches should therefore get the highest priority. Furthermore, our results show that increasing the size and the number of tree species in a patch are the most effective measures to increase the (plant) species richness in the patch.

4.3. Joint comparison of economic and ecological results

Our study shows contrasting results between the economic analysis of social preferences and the outcomes of ecological analysis.

The results of the ecological analysis pointed out that increasing tree species richness is more important than establishing a shrub layer or creating a heterogeneous canopy structure to increase the total herb species richness in the patch. However, this attribute did not achieve significant willingness to pay estimates. As we mentioned above, we hypothesize that this may be related to the deciduous character of these patches and the reduced impact of this change on the aesthetic experience of the respondents.

The ecological analyses show that increasing the forest area by enlarging the forest patches has clearly a large effect on the richness of herb species in general and on forest herb species in particular. This species-area relationship is well-established in the ecological literature (e.g. Rosenzweig, 1995). The social preferences are aligned with the ecological findings in terms of prioritization of area enlargement in the bocage region, with WTP estimates being higher for this measure among bocage respondents (16.44 €/individual for bocage sample vs. 11.46 €/individual in the open field region). These results show that social preferences and ecological effectiveness do differ between landscapes that represent different degrees of biodiversity conservation, with the same measure producing different social gains depending on where it is applied. Furthermore, we see that preferences of open field respondents for increasing biodiversity content with limited increase in forest area proves difficult to attain based on the evidence provided by ecological outcomes.

Despite the fact that herbaceous richness is a stable indicator to assess the ecological status of forest ecosystems, our study shows that this is not necessarily appreciated by the general public. While other studies show significant estimates for improvement of species richness, these corresponded to threaten ones (e.g. Campbell et al., 2014; Domínguez-Torreiro and Soliño, 2011); the fact that our study assesses species in general (and not specifically threatened) may have less compelled respondents to act (Jacobsen et al., 2008). In addition, and differently from these studies rather than pooling together all the species in a more general fashion, we let respondents express their priorities (and trade-offs) among three different taxonomic groups. These differences in the design of our study may contribute to explain the disparities found with previous studies.

4.4. Policy implications

Attitudes and perceptions of stakeholders over small forest fragments and surrounding agricultural matrix may influence forest policy implementation; therefore effective policy design requires understanding of stakeholders' perception of ecosystem services provided by those forest patches (Lamarque et al., 2011).

Policy makers have to contrast economic information with ecological effectiveness, finding a balance between them when these signal differing paths of action. In this study, higher welfare gains are obtained for OF respondents compared to B respondents with regards to improving the condition of existing patches (i.e. improvement in the number of butterfly or bird species and structural improvement other than tree species). These are coherent with the neoclassical rationality of diminishing marginal utility gains (Horowitz et al., 2007), i.e. dwellers where the landscape is perceived as more “degraded” attach more value to biodiversity and structural diversity than dwellers in places comparatively better-off on these terms. The ecological data support that the richness in the OF landscapes is lower than in the B (Fig. 3). Should the utility gain be the only criteria to consider, the more degraded areas should receive most of the funds to restore their ecological quality. However, ethical issues of fairness may arise as those with more potential to increase biodiversity are likely those who “polluted” more in the past through intensifying agricultural land-use (Wunder, 2007); additionally, some studies point out that nature conservation measures are needed even in B type landscapes to halt strong species loss (Van Calster et al., 2008).

From an ecological point of view, a minimum forest patch area may be required to overcome a tipping point which avoids irreversibility in terms of biodiversity degradation; hence policies could establish such threshold (Fisher et al., 2008) and introduce incentives from there onwards. This illustrates the need for a pluridisciplinar assessment of ecosystems where a diversity of criteria are considered in decision-making processes (Berkes et al., 2008; Filotas et al., 2014).

5. Conclusions

This work conducted ecologic and welfare economic assessments concomitantly, thus allowing a joint comparison of the key attributes that play a decisive role in determining biodiversity patterns and their contribution to shape social preferences for these forest patches.

This scope shows disparities and similarities between economic and ecological criteria, signalling the challenges that decision-making processes related to ecosystem management have to face to embrace the complexity of socio-ecological interactions. The lack of social acceptability or, alternatively, the reduction of biodiversity levels are the risks that management would face should it be solely based either on ecological variables or on social preferences, respectively.

Acknowledgments

This research was funded by the ERA-Net BiodivERsA project smallFOREST, with the national funders ANR (France), MINECO (Spain), DFG/DLR (Germany) and BELSPO (Belgium) as part of the 2011 BiodivERsA call for research proposals. This paper is also a contribution by KV, MS and JBJ to the Cost Action FP1206 "EuMIXFOR" (*European mixed forests - Integrating Scientific Knowledge in Sustainable Forest Management*).We acknowledge Marina Autin, Déborah Closset-Kopp, Hélène Horen, Thomas Jazeix, Ludmilla Martin, Matthieu Rubrecht in the French case study, and Laurien Spruyt and Fien De Rudder in the Belgium case study for their help in conducting the interviews. The authors would like to thank Anders Busse Nielsen for kindly granting them with access to the Tree Library containing his forest development type illustrations that have been used in preparing the choice sets to convey stand structure attributes to the respondents. Last but not least, the authors would like to thank all the anonymous surveyed respondents for their participation.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2017.11.190.

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