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On the Interest of Litter-Dwelling Invertebrates to Assess Silvicultural Impact on Forest Biodiversity

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Abstract

In the actual context of urgent need of biodiversity indicators to assess forest ecosystems, the use of insects and more generally invertebrates remains infrequent whilst they constitute the major part of biodiversity.

Our paper, focusing on litter-dwelling invertebrates (*Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda*) at the stand level, come within the framework of the validation of indicators of forest biodiversity. So our research is attempting to partly fill in the gaps in our knowledge about correlations between indicators (e.g. coarse woody debris, main tree composition, stand structure and stand age) and litter-dwelling invertebrates species richness. The correlations between the groups under investigation and with plant richness are then considered. These correlations appear to be low so the reliability of one taxonomic group as indicator should always be cautiously tested.

Taking into account bioindicator criteria, the indicator ability of the four group studied is discussed. *Staphylinidae* seem to constitute the best indicators of forest management and naturalness in landscapes with long human disturbance history.

Keywords: Carabidae; Staphylinidae; Diplopoda; Chilopoda; silviculture; biodiversity indicator.

1. Introduction

In the context of the current biodiversity crisis and the subsequent development of political, legal and management instruments in favour of biodiversity, we urgently need reliable

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indicators of biodiversity. Such indicators play a major role in monitoring biodiversity, identifying, targeting and helping prioritise the types of action to be taken in forest ecosystems and in measuring and assessing the results of the implementation of such actions (Noss 1990; Larsson and Esteban 2000; Watt 2003).

In a long-term perspective, it is necessary for a standardised system of measurable attributes or indicators of forest biodiversity to be implemented in a global assessment of state and trends in biological diversity (Noss 1990; Larsson 2001).

But conceptual problems arise, biodiversity have various meanings for different people. The ways in which biodiversity is considered are numerous and depend frequently on the speciality of each scientist (Noss 1990; Rainio and Niemelä 2003).

A contradiction occurs concerning bioindicators. While insects and more generally invertebrates constitute the major part of the biodiversity and particularly in forest ecosystems (Figure 1), they are quite rarely used as indicators. This statement is easily explained by the concern for saving money, simplicity and rapidity. This is why forest structure and plants (taxonomically well described and easily identified in the field) are frequently used to estimate forest biodiversity. However, little is known about the correlations between plant richness and invertebrate richness, moreover there is some evidence that flowering plant diversity does not constitute an accurate indicator of the species richness of invertebrates (Oliver and Beattie 1993; Larsson and Esteban 2000).

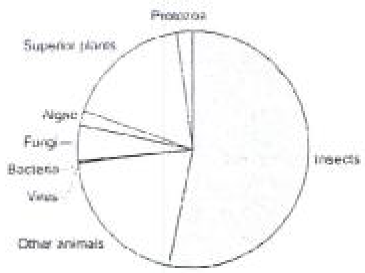
The quality and reliability of the chosen indicators should be scientifically tested by firstly defining without fog the objectives and the corresponding biodiversity aspects or entities. In a time-consuming second step, a number of habitats have to be sampled as thoroughly as possible with regard to the aims defined. The third step consists in testing the correlations between our choice of easily measurable indicators and the habitat samplings done in the second step. Most studies claiming to measure or indicate biodiversity assume that the group of organisms they investigate is somehow representative of biodiversity. However, in only very few cases has the correlation between a group or several groups of species with a more or less representative sample of all organisms been measured and published (Duelli and Obrist 2003). So although there is a wealth of indicators to choose from, most have been poorly tested and require rigorous validation in order to be used and interpreted with confidence (Noss 1999; Larsson 2001). Our research is trying to partly fill in these gaps. Besides bioindicator criteria, sensitivity to forest management and correlations between the groups studied and with vegetation species richness brought valuable information about the bioindicator power of litter-dwelling invertebrates (*Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda*).

2. Methods

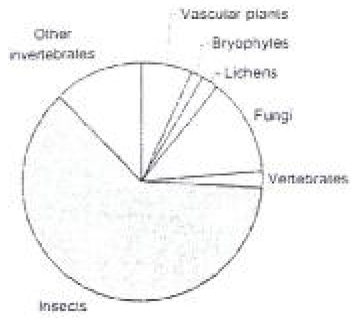
2.1 Study region, sampling design and selection of plots

The study (du Bus de Warnaffe 2002) took place in the natural region of the Belgian Ardennes (about 5000 km²), situated between the cities of Namur and Luxembourg. The Ardennes are mostly composed of pastures and woodlands, partially transformed into commercial conifer stands during the last 150 years (Devillez and Delhaise 1991). This region is characterised by a humid sub-mountane climate, a hilly relief and loamy acidic soils.

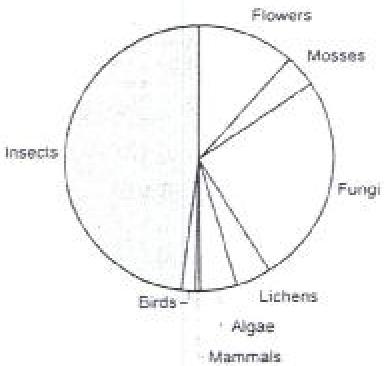
The plots were chosen in order to minimise the variation of climate and soil, and according to three categorical management variables: the *structure* of the forest, the *composition* of the canopy and the *stage* reached in the silvicultural cycle. The altitude ranged from 320 to 600 m, the mean annual rainfall from 830 to 1170 mm yr^{-1} and the mean annual temperatures from 6.7



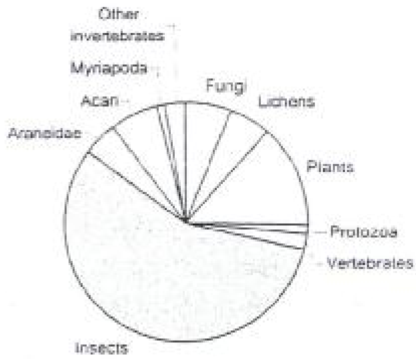
Insects are about 1 million species of which circa 400 000 *Coleoptera* (Wilson 1993).



Bialowicza forest (Poland and Bielorussia). Insects are about 9282 species of which 2884 *Coleoptera* (Falinski 1991; Gutowski and Jaroszewicz 2001).



Close-to-nature Atlantic beech forest of Fontainebleau (Central France). Insects are about 5600 species of which circa 3200 *Coleoptera* (ONF 2003).



Close-to-nature Mediterranean mountain beech forest of Massane and associated habitats (Pyrénées, France). Insects are about 2776 species of which 1434 *Coleoptera* (Travé et al. 1999).

Figure 1. Proportion of known species richness in the main living groups.

to 9.2 °C (Weissen et al. 1994). All the study plots were on flat or very slightly sloping ground of acidic brown and moderately dry soils (*Dystric cambisol*: FAO 1990), which were very similar in terms of water and nutrient availability. All stands were forests for at least 150 years. Plots were all situated in large forests and at least 100 m from the nearest field or meadow.

The *structure*, as determined by the mean size of patches (homogeneous stand) in the forest, was evaluated by Geographical Information System (G.I.S. Star-Carto) on aerial photographs. Three classes were distinguished: even-aged (E): final tree exploitation by large clear-cuttings (> 2 ha), group (G): final tree exploitation by medium-sized clear-cuttings (0.2–0.5 ha), and uneven-aged (U): final tree exploitation by small clear-cuttings (< 0.2 ha).

The *composition* classes were defined by the local cover of the tree species (0.04 ha): more than 90% of the area covered by beech (B: *Fagus sylvatica* L.), by oaks (O: *Quercus petraea*

Table 1. Plot number per habitat type which are combinations of the structure (first initial), composition (second initial) and stage reached in the silvicultural cycle (see 2. Methods). Except the close-to-nature stands in brackets (r), all the plots are intensively managed.

Structure - composition	EC	EB	EO	GM	UC	UB	Total
Stage 1	8	6	6	6	5	10	41
Stage 2	8	6	22 (r =16)	6	7	10	59
Stage 3	8	6	8	6	6	24 (r =14)	56
Stage 4	-	4 (r =4)	-	-	-	-	4
Total number of plots	24	22	36	18	18	44	162

Table 2. Total number of species and individuals trapped in the 162 plots.

	Carabidae	Staphylinidae	Diplopoda	Chilopoda
Number of species	80	136	19	25
Number of individuals	43 658	13 608	4334	3293

(Mattme.) Liebl. and *Quercus robur* L.), by spruce and Douglas fir (C; *Picea abies* (L.) Karst and *Pseudotsuga menziesii* (Mirb.) Franco) or by a mixing of the previous species (M).

Three *stages* were defined for each combination of structure and composition: regeneration (stage 1: trees aged 3–10 years); medium-aged stand (stage 2: trees aged 20–40 years for conifer stands and 30–60 years for beech and oak stands), mature stand (stage 3: trees aged 50–80 years for conifer stands and 80–140 years for beech and oak stands) and overmature stand (stage 4: trees aged > 150 years for beech stands).

We also distinguished between intensively managed stands (m) and close-to-nature stands (r; reference stands: indigenous tree species, native provenance, natural regeneration, dead wood abundance and forest continuity in time).

The 21 classes defined by combining structure, composition and stage were called *habitat types*. In all, 162 plots were selected. The number of plots by class is given in Table 1.

2.2 Species data

We used 8.5 cm diameter × 17 cm deep pitfall traps with 5% formaldehyde (over-saturated salty solution was used in close-to-nature stands) to collect ground-dwelling arthropods (Dufrêne 1988). In each plot, 3 pitfalls were placed in a triangle of 3 m base (Desender et al. 1999), and emptied monthly (Heliölä et al. 2001). We sampled the traps for seven months (Benest and Cancela da Fonseca 1980; Düldge 1994), from 10 April to 5 November 1999 (384 traps in intensively managed stands) and from 1 April to 31 October 2002 (84 traps in close-to-nature stands). Pitfall traps are known to gather valuable information on activity and relative abundance of various groups of ground-dwelling arthropods (e.g. Dufrêne 1988; Branquart et al. 1995; Rainio and Niemelä 2003). Among all the groups captured, only the most representative and relevant taxa were sorted, identified and analysed: *Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda* representing an important part of the litter-dwelling invertebrates. The classification of the 162 sample plots in the 21 habitat types allowed us to compute mean species richness and cumulative number of species trapped for each habitat type.

Table 3. Main results for the *Carabidae*, *Staphylinidae* and *Diplopoda* for each habitat type (n = plot number; N = mean individual number; S = mean species richness and Stot = total number of species trapped per habitat type).

Habitat type		n	*Carabidae* N	*Carabidae* S	*Carabidae* Stot	*Staphylinidae* N	*Staphylinidae* S	*Staphylinidae* Stot	*Diplopoda* N	*Diplopoda* S	*Diplopoda* Stot
Beech forest	EB-1m	6	303.0	20.5	43	137.8	11.0	32	30.0	5.8	11
	EB-2m	6	250.5	8.5	19	36.0	7.3	19	7.7	2.2	7
	EB-3m	6	225.0	8.3	18	30.8	6.7	20	14.5	2.3	6
	EB-4r	4	87.8	9.0	16	70.0	19.0	42	4.0	1.0	1
	UB-1m	10	272.0	13.8	36	33.8	10.7	35	24.6	4.2	9
	UB-2m	10	354.4	8.0	23	65.7	9.0	25	21.9	3.6	8
	UB-3m	10	355.3	8.6	22	50.1	8.2	26	11.0	3.0	7
	UB-3r	14	240.9	11.2	32	56.5	12.0	64	74.1	5.5	13
Mixed forest	GM-1m	6	104.0	15.3	36	52.5	12.2	33	25.8	4.5	9
	GM-2m	6	148.5	9.3	23	65.0	10.3	29	28.2	5.2	11
	GM-3m	6	223.0	8.7	19	53.7	9.3	23	17.2	3.7	8
Coniferous forest	EC-1m	8	220.0	17.4	40	109.6	14.6	42	57.9	4.5	12
	EC-2m	8	278.5	9.5	23	72.3	9.3	29	14.3	4.0	10
	EC-3m	8	333.6	9.5	18	60.3	8.1	23	5.9	3.1	8
	UC-1m	5	63.4	8.6	20	40.6	11.0	27	44.2	5.4	10
	UC-2m	7	227.6	7.9	16	207.7	9.0	27	25.0	3.9	10
	UC-3m	6	276.5	7.7	17	49.8	9.5	28	10.7	3.8	7
Oak forest	EO-1m	6	147.8	14.2	33	49.5	8.8	32	29.8	3.5	7
	EO-2m	6	289.3	11.3	23	202.5	6.3	14	49.2	5.2	8
	EO-2r	16	393.3	14.4	47	77.8	15.7	68	26.1	4.4	13
	EO-3m	8	431.4	12.9	29	267.0	4.9	21	41.9	4.6	11

EB = even-aged beech stand
UB = uneven-aged beech stand
EO = even-aged oak stand
GM = group mixed stand
EC = even-aged coniferous stand
UC = uneven-aged coniferous stand

1 = regeneration stand
2 = growing stand
3 = mature stand
4 = overmature stand
r = close-to-nature stand
m = intensively managed stand

Vascular plant, bush and tree species richness was obtained for the 162 plots by sampling vegetation on 0.04 ha according to Braun-Blanquet method (Braun-Blanquet 1951).

Finally using the SAS package, we computed the Spearman rank correlations between species richness of vascular plants, bushes, trees, *Carabidae, Staphylinidae, Chilopoda* and *Diplopoda* (SAS 2000).

3. Results

The paired comparisons (Table 3) between the same habitat type show that close-to-nature forests shelter a higher species richness for the *Coleoptera Carabidae* (except for Stot in EB) and *Staphylinidae* in comparison with intensively managed forests. The results are less clear for *Diplopoda* whose species richness is quite poor. The *Chilopoda* results not presented here gave similar statistics to *Diplopoda*.

Whilst the total number of species trapped are quite similar between *Carabidae* and *Staphylinidae*, the difference between close-to-nature and intensively managed forests is

Table 4. Spearman rank correlations between species richness of vascular plants, bushes, trees, *Carabidae* (Car), *Staphylinidae* (Staph), *Chilopoda* (Chilo) and *Diplopoda* (Diplo) (*: 0.05<p<0.01; **: 0.01<p<0.001; ***: 0.001<p<0.0001 ; ****: p<0.0001).

	Plant	Bush	Tree	Car	Staph	Chilo	Diplo
Plant	-	0.50****	0.04	0.26***	0.28***	-0.14	0.06
Bush		-	-0.07	0.05	0.003	0.06	0.21***
Tree			-	-0.25**	0.02	0.05	0.12
Car				-	0.26***	-0.19*	0.17*
Staph					-	-0.37****	-0.06
Chilo						-	0.15
Diplo							-

much higher for *Staphylinidae* than for *Carabidae*. This statement is supported by the fact that the trappings in close-to-nature stands brought nine *Staphylinidae* species new for Belgium.

Examining now the correlations between the species richness of the groups under investigation (Table 4), we observe that these relations are in general low and can be positive, absent or even negative. This is also true concerning the correlations between the vegetation richness (vascular plants, bushes or trees) and the invertebrate richness. Yet, the species richness of *Staphylinidae* and *Carabidae* are slightly correlated (0.26, p=0.0008) and the species richness of *Staphylinidae* and *Chilopoda* are negatively correlated (-0.37, p<0.0001). The correlations between the groups under investigation and vegetation are low. *Carabidae* (0.26, p=0.0006) and *Staphylinidae* (0.28, p=0.0003) present significant correlation with vascular plants, *Diplopoda* with bushes (0.21, p=0.0062) and *Carabidae* are negatively correlated with trees (-0.25, p=0.0014). These results are more or less similar to the figures obtained by Saetersdal et al. (2003).

4. Discussion

According to our results comparing close-to-nature stands to intensively managed stands for the same habitat types, it appears that forest management has a negative impact on the species richness of litter-dwelling invertebrates.

The poor *Diplopoda* and *Chilopoda* species richness observed is mainly due to acidic soil conditions, altitude and harsh climate, at least above 500 m (Kime pers. communication). This statement limits the value of our results for analysing their indicative power.

Carabidae communities have been seriously altered in Belgian forests (du Bus de Warnaffe and Lebrun in press). Large brachypteran forest stenotopic species have disappeared or have seen their population decreasing. Strict forest species such as *Carabus intricatus* or *C. glabratus* have disappeared in Belgian forests and *Abax carinatus* has seen its population diminishing drastically. In fact, only the species well adapted to silvicultural perturbations could still be present in Ardennes forests, intensively managed for centuries. Due to this situation, we have to use cautiously naturalness differences between our Belgian forests with this group. The *Carabidae* species richness, higher in regeneration stage (Table 3), is due to the colonisation of large clear-cuttings by open-field species (du Bus de Warnaffe and Lebrun in press). So it appears clearly that the indication given by the species richness is insufficient.

It is always important to examine the list of species (red-listed, rare and endangered species, stenotopic species, etc.).

Of the four groups, *Staphylinidae* constitutes the best indicator of the main forest management types but also of forest naturalness. This family is very common in natural, semi-natural and managed forest ecosystems (Bohac 1999).

While the majority of species are known as non-specific predators, a significant proportion of species possess narrow ecological requirements of prime importance to assess the naturalness of forest ecosystems (i.e. *mycetophagous*, *fungicolous*, *myrmecophilous* or *saproxylophagous*, etc.). Unfortunately this family still remains insufficiently known (Koch 1989; Bohac 1999). In any case, *Staphylinidae* constitute a very promising group not only as bioindicators but also for future ecological studies.

As stated by Larsson (2001), there is a need for reference values and critical thresholds for forest biodiversity. A reconstruction of reference situations of forest biodiversity (e.g. near-native state, pre-industrial state) expressed through indicators values would be of practical use for setting biodiversity targets (Larsson 2001). We are trying to approach this reference i.e. the state of *Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda* in the natural undisturbed community in order to possibly place our stands on a naturalness scale and to distinguish the actual "anthropic biodiversity" versus "natural biodiversity" under low human influence.

We intend to realise this approach in a near future by combining data:

- from natural or near-natural European forest ecosystems in, as far as possible, similar ecological conditions;
- on "strict forest species" and more precisely on "forest naturalness indicators".

Although the indicator species approach is opened to criticism (questionable assumptions, methodological difficulties, biased applications, etc.), species are often more tangible and easy to study than communities, landscapes or genes (Noss 1990). Noss (1999) proposed seven kinds of species that might make good targets for monitoring supposing that these most sensitive focal species (or group of species) would presumably constitute umbrella species (area-limited, dispersal-limited, resource-limited, process-limited, keystone, narrow endemic and finally special cases which gather species important in the forest ecoregion that do not fall within one of the previous categories).

The bioindicators, rigorously tested, must be measurable and repeatable surrogates for biodiversity which should ideally be:

1. well-known in its taxonomy and biology;
2. easy and cost-effective to measure, collect, identify and/or calculate;
3. widely applicable and then distributed over a broad geographical area
4. distributed over a large range of habitats with species that are pledged to specialised habitat in order to be sufficiently sensitive to provide an early warning of change;
5. capable of providing a continuous assessment over a wide range of stress and also able to differentiate between natural cycles or trends and those induced by anthropogenic stress (e.g. Noss 1990; Dajoz 1996; Rainio and Niemelä 2003).

Because no single indicator will possess all of these desirable properties, a set of complementary indicators is required (Noss 1990; Larsson and Esteban 2000). Table 5 examines the characteristics of the litter-dwelling groups under investigation in regard to the indicator desirable properties given above.

Whilst, according to Table 5, *Carabidae* seem to gather the best results concerning the bioindicator criteria, *Staphylinidae* appear to be more sensitive and then to bring more valuable information in landscapes with long human disturbance history.

Table 5. Indicator properties of the groups under investigation (Car: *Carabidae*; Staph: *Staphylinidae*; Diplo: *Diplopoda*; Chilo: *Chilopoda*).

Desirable properties	Bioindicator criteria	Car	Staph	Diplo	Chilo
1	Identification key	++	+/-	++	++
1	Checklists	++	+/-	++	++
1	Experts	++	+	+	+
1	Ecological information	+++	+/-	+	+
1	Chorological data	++	+/-	+	+
2	Easy to identify	+	-	+	+/-
2	Easy to sample (standardised)	+	-	+	+
2	Cost-effective	+/-	-	+/-	+/-
3	Distribution	broad	broad	broad	broad
3	Widely applicable	+++	+++	+++	+++
4	Habitat range	++	+++	++	++
4	Specialised species	++	+++	+	+
4	Sensitivity	++	+++	++	++
4	Forest naturalness indicators	+	++	-	-
4	Indicators of forest continuity	+/-	+	-	-
5	Continuous assessment	+++	+++	+++	+++
5	Natural >< anthropogenic	+	+++	+/-	+/-

Besides species or species group, biodiversity indicators can be habitat variables (structural and/or functional variables such as diameter and age class distributions, landscape pattern using remote sensing, etc.) (Noss 1999; Saetersdal et al. 2003). In many landscapes however, human land-use indicators (both structural and functional: e.g. deforestation rate, road density, fragmentation or edge index, etc.) may be the most critical variables for tracking the status of biodiversity (Noss 1990). Such habitat variables assumed or ideally proved to be important to the species (habitat suitability indicators) can usefully complete species richness and sometimes even replace the daunting samplings necessary to estimate this richness. Dead wood is a structural element critical to forest biodiversity and in particular to saproxylophagous species which represent one of the most important compartment of biodiversity in forests (e.g. Speight 1989; Noss 1990; Samuelsson 1994; Martikainen 2000). So the assessment of dead wood cannot be ignored in any forest biodiversity evaluation. Unfortunately, the presence of suitable habitat is no guarantee that the species of interest are present. Thus monitoring both habitat variables and species seem to be essential in most cases (Noss 1990; Saetersdal et al. 2003).

5. Conclusions and prospects

Two major results emerge from our study. First, of the four litter-dwelling groups investigated, *Staphylinidae* seem to constitute the best indicators of forest management and naturalness in landscapes with long human disturbance history and a fortiori in the Belgian conditions. Yet this group has the drawbacks of remaining insufficiently known and difficult to identify. And second, our analysis show that the frequently assumed correlations between the richness of different groups is not so evident. This is the case notably for plant richness which appears to be poorly correlated to the *Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda*.

In biodiversity assessment matters, there is a consensus for an interdisciplinary approach including scientifically tested biological and habitat indicators. However, we should avoid the situation where the number of indicators required becomes an obstacle for an operational indicator system (Larsson and Esteban 2000). So in order to be applicable on the field, the set of indicators required should remain easy to use by field technicians.

Although the wealth of forest biodiversity indicators, the choice of reliable ones remains critical and unresolved. To select them, the major task remaining is to test and validate our ostensible indicators to check what they are really telling us (Noss 1999; Nilsson et al. 2001). Still of little use, the most species rich organism group, the invertebrates and in particular insects, should be included in any serious biodiversity assessment method (Dajoz 1996; Nilsson et al. 2001). To reduce the money, time and efforts required to collect and identify such complex group, two solutions exist. The first one, the "rapid biodiversity assessment" using "recognisable taxonomic units" was successfully tested for some groups. This method consists in collecting within a few selected weeks in a standardised trap combination and just considering the level of morphospecies, i.e. taxa that are readily separable by morphological differences obvious to non-specialists (Oliver and Beattie 1993, 1996). And the second one consists in finding suitable indicators (habitat variables, other easiest groups) providing a good assessment of the invertebrates (e.g. dead wood characteristics and abundance for xylophagous insects).

For forests biodiversity assessments, the interdisciplinary approach could follow the steps listed hereafter for which the indicators must be specified depending on the objectives and the scale under investigation:

1. general description of the area (topography, soil, geology, climate, landscape matrix, phytosociology and habitat types);
2. structure of the forest (dendrometric approach at the stand-level and remote sensing approach at the landscape level);
3. complementary and relevant bioindicators (depending on the scale, the habitat type and its situation) respecting the bioindicator criteria (at least trees, plants, lichens, lignicolous fungi, some insect groups, other invertebrates and possibly mosses and some mammal or bird species respecting the criteria developed by Noss 1999);
4. complementary and reliable habitat variables (a broad example of these variables is given by Noss 1999).

Finally, we have to bear in mind that monitoring, research and land planning should be in continual interplay, with information from one always informing the others in order to reach an adaptive management. But the time lags in the response of populations to habitat degradation suggest that when the decline is detected, it may be too late to make the necessary changes in management (Noss 1999). So the best way remains to reproduce as much as possible the characteristics of close-to-nature ecosystems (e.g. disturbance regime, etc.). But the lack of reference is sorely felt, hence the prime importance of natural woodlands and strict forest reserves which can be studied as references to settle the indicators of forest biodiversity and their thresholds.

To complete this paper, our future prospects will be:

1. to identify the silvicultural reasons explaining the different communities of the groups studied prevailing under different forest habitat types;
2. to identify the "forest naturalness indicators" allowing us to classify forests on a naturalness scale;
3. to bring to the fore simple and reliable silvicultural indicators of the identified management influences in order to avoid, if possible, the tedious assessment of the invertebrates;
4. and finally to draw biodiversity-friendly silvicultural recommendations.

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