Changes in the distribution of carabid beetles in Belgium revisited: Have we halted the diversity loss?

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# a b s t r a c t

Twenty years ago, [Desender and Turin (1989)](#_bookmark6) analysed the changes in the composition of carabid beetles in four NW European countries between the periods <1950 and 1950–1985. Recently, a new distribution atlas of carabid beetles in Belgium was compiled using data collected during the period 1986–2008. In the light of the Countdown2010 target of halting the loss of biodiversity, we used these new data to test whether or not previously observed trends were altered. Since 1950, 46 species were no longer recorded in Belgium and seven species were added to the Belgian fauna. By relating the changes in distribution area to ecological and life history traits as well as to conservation priorities of the species, we examined which species characteristics were associated with the strongest changes in distribution. Comparing the period before 1950 with the period 1950–1985 showed that species from nutrient-poor dry biotopes and heathlands, threatened, rare and big species declined. Generalists, non-threatened species, species with a pan-European distribution range, species in the centre of their distribution range and common species, on the other hand, increased. From the period 1950–1985 to 1986–2008, mainly macropterous species, both rare and very common species and big species decreased, while generalists, dimorphic species, species with a pan-European distribution range and species that were already common in the second period increased. For the conservation of carabid beetles in a strongly industrialised and highly fragmented NW European landscape, we propose actions on two levels: ﬁrst, the protection and adequate manage- ment of high quality biotopes, especially nutrient-poor grasslands and heathlands, in large core areas for specialist species and second, the creation and/or restoration of a ‘matrix’ that facilitates the exchange of individuals between core areas for the conservation of both generalist and specialist species.

1. Introduction

Biodiversity is declining rapidly, especially in strongly indus- trialised regions. This is mainly due to changes in land use (biotope destruction, fragmentation, eutrophication, etc. – [Warren et al.,](#_bookmark41) [2001](#_bookmark41)), to climate change ([C.D. Thomas et al., 2004; J.A. Thomas](#_bookmark27) [et al., 2004](#_bookmark27)) and to the introduction and expansion of invasive spe- cies ([Crowl et al., 2008](#_bookmark19)). To tackle this biodiversity crisis, policy makers have set global targets (‘‘*reducing* the loss of biodiversity”) and even more ambitious European targets (‘‘*halting* the loss of bio- diversity”) also known as the Countdown2010 target. Conse- quently, we need ‘‘indicators” that are suitable for measuring the

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1 Konjev Desender died during the writing of this publication and we sincerely

thank him for all the knowledge and enthusiasm he passed onto us and many other coleopterists in Belgium and elsewhere during the many years of collaboration.

progress towards these targets ([Balmford et al., 2005](#_bookmark10)). One of the headline indicators selected by the Convention on Biological Diver- sity (CBD) is the trend in abundance and distribution of selected species ([Mace and Baillie, 2007](#_bookmark28)). These are often species for which information is readily available (e.g., birds, mammals, plants), but they represent only a small fraction of biodiversity. Invertebrates are rarely selected as ‘‘indicator” species despite their species-rich- ness and excellent early-warning capacities: they react locally and, due to their relatively short life cycles, more rapidly than many long-lived animals or plants ([Thomas, 1994; J.A. Thomas et al.,](#_bookmark27) [2004; Samways, 2005](#_bookmark27)).

Carabid beetles are very diverse and belong to the best studied invertebrates in the world, together with butterﬂies and dragon- ﬂies. Research on distribution and population ecology and conser- vation of carabid beetles has a long tradition in Europe ([Lindroth,](#_bookmark28) [1945; den Boer, 1970; Thiele, 1977; Lindroth, 1985, 1986; Erwin](#_bookmark28) [et al., 1979; Desender et al., 1994a](#_bookmark28)). In Belgium, collecting of cara- bid beetles started halfway the 19th century, when coleopterists mainly focused on the large and conspicuous species of the genera

*Carabus* and *Calosoma* ([Desender et al., 1994b](#_bookmark6)). Carabid beetles occur in a wide variety of biotopes, are relatively easy to identify and can be collected in a standardised way by pitfalls. They are often considered as good bio-indicators ([Dufrêne and Legendre,](#_bookmark11) [1997](#_bookmark11)) and as useful evaluation tools for nature conservation pur- poses ([Refseth, 1980; Rainio and Niemelä, 2003; Desender et al.,](#_bookmark42) [2007](#_bookmark42)). Because of their more restricted niche, compared to birds or mammals for example, and their occurrence in a wide variety of biotopes (e.g., mud ﬂats, bare sand areas, ancient woodlands), conservation targets for carabid beetles and other invertebrates can be considered as complementary to those for mammals, birds and/or plants ([Maes and Van Dyck, 2005](#_bookmark28)).

Conservation actions are usually undertaken on a national or on a regional level and therefore it is necessary to have local indica- tors that assess the Countdown2010 target ([Failing and Gregory,](#_bookmark12) [2003; Mace and Baillie, 2007](#_bookmark12)). Due to a high level of nitrogen depo- sition, intensive agriculture and the high degree of fragmentation of natural areas, Belgium (especially the northern part, Flanders) is one of the regions in Europe that has lost a considerable part of its biodiversity (e.g., carabid beetles – [Desender and Turin,](#_bookmark6) [1989](#_bookmark6); butterﬂies – [Maes and Van Dyck, 2001](#_bookmark28); vascular plants – [Van Landuyt et al., 2008](#_bookmark29)). In this paper, we calculated changes over three periods in the size of the distribution area of individual cara- bid species (measured as the number of survey grid cells in which a species occurred). The basic data were taken from a published comparison between two periods (<1950 *vs.* 1950–1985; [Desender](#_bookmark6) [and Turin, 1989](#_bookmark6)) and complemented with distribution data from a third period (1986–2008; [Desender et al., 2008b](#_bookmark13)). We ﬁrst examine to what extent the previously detected loss in carabid beetles has been halted. Second, we test whether changes in distribution area can be related to ecological and life history traits or to a species’ threat status. Finally, we discuss whether progress was made in the conservation of carabid beetles in the light of the Count- down2010 target and propose possible ideas for conservation to preserve the most declining species.

1. Material and methods
	1. *Study area*

Belgium is a strongly industrialised NW European country with a high human population density (341 inhabitants/km2, [Dumortier](#_bookmark14) [et al., 2007](#_bookmark14)) and, consequently, intense pressure on nature ([OECD,](#_bookmark34)

[1998](#_bookmark34)). The landscape and topography differ considerably between the two administrative regions of Belgium: Flanders and Wallonia. Flanders, the northern part, is a lowland zone (mean eleva- tion = 38 m) and has only a limited total area of nature reserves (2.5% of the territory, [Van Goethem, 2001](#_bookmark30)) and forest (8% – [CEC,](#_bookmark15) [1994](#_bookmark15)). Wallonia, the southern part and comparatively an upland region (mean elevation = 310 m) has a smaller area of nature re- serves (0.6% of the territory, [Van Goethem, 2001](#_bookmark30)), but has a consid- erably higher amount of woodlands, equally distributed over coniferous and deciduous woodlands (31%, [CEC, 1994](#_bookmark15)).

* 1. *Study species and data*

Carabid or ground beetles (Coleoptera, Carabidae) are terrestrial invertebrates that occur in a wide variety of biotopes. Nowadays, carabids are usually collected with pitfalls, but older and also a part of the recent records were collected by hand or by net-sweeping ([Southwood, 1978](#_bookmark27)). All records (i.e., a species on a given date at a given site) were attributed to 5 x 5 km grid cells of the UTM (Uni- versal Transverse Mercator) projection, hereafter called grid cells. In total, about 200,000 records were collated of 404 species from a total of 1194 surveyed grid cells (i.e., 87% of the total number of grid cells in Belgium). With this dataset, a new distribution atlas for Belgium was compiled using all the recently collected data by

162 volunteer and professional coleopterists ([Desender et al.,](#_bookmark13) [2008b](#_bookmark13)). Nomenclature of the species is according to [Desender](#_bookmark13) [et al. (2008b)](#_bookmark13). The mapping intensity per ﬁve year period since 1830 is given in [Fig. 1](#_bookmark0). From this distribution atlas, we compiled a dataset with ‘species–period–grid cell’ as units, resulting in *ca.* 105,000 condensed records ([Table 1](#_bookmark1)).

The total number and the grid cells surveyed in each period are given in [Table 1](#_bookmark1) and [Fig. 2](#_bookmark2)a. We restricted our analysis to the 360 species (89% of all Belgian species, cf. [Eyre et al., 2005](#_bookmark16)) that oc- curred in at least ﬁve grid cells in at least one of the periods. For further analysis, we only used grid cells that have more than 25% of their area in Belgium (*N* = 1292) and that were relatively well surveyed (i.e., at least ﬁve species in every period). This reduced the data set for analysis to about 65,000 condensed records from 293 grid cells (23% of all Belgian grid cells, [Table 1](#_bookmark1) and [Fig. 2](#_bookmark2)b).

* 1. *Analysis*

[Desender and Turin (1989)](#_bookmark6) compared changes in species com- position between two periods: before 1950 and 1950–1985. Here,

**12000**

<1950

1950-1985

1986-2008

**10000**

**8000**

**6000**

**4000**

**2000**

**0**

Fig. 1. Number of records per period of 5 years in Belgium since 1830. The different colours indicate the different periods used in the analysis.

Table 1

The number of all species, species no longer recorded and new species observed in the three investigated periods in Belgium (a). The total number of grid cells with observations, condensed records (i.e., species–period–grid cell) and the number of condensed records used in the analysis in the three investigated periods in Belgium (b).

compared to the previous period

Number of new species compared to the previous period

2

17

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | <1950 | 1950–1985 | 1986–2008 | Total |
| *a* |  |  |  |  |
| Number of species | 397 | 353 | 360 | 404 |
| Number of species no longer recorded | 46 | 10 |  |  |

|  |  |
| --- | --- |
| *b* |  |
| Number of grid cells with observations | 852 | 1016 | 892 | 1194 |
| Number of condensed records | 39,797 | 29,136 | 36,316 |  |
| Number of records used in the analysis | 21,482 | 19,747 | 23,289 |  |

we compared the same periods as in [Desender and Turin (1989)](#_bookmark6) and add a third period: 1986–2008. To compare the trend between the periods <1950 and 1950–1985 with those between the periods 1950–1985 and 1986–2008, we applied the same methodology as in [Desender and Turin (1989)](#_bookmark6). They used a G-test that compared the number of grid cells occupied by a species in two periods with the number of grid cells summed for all other species in those peri- ods. We considered species as signiﬁcantly decreasing or increas- ing in distribution area when the *G*-test had a *p*-value <0.10. Species where changes in distribution area were non-signiﬁcant (*p* > 0.10) were considered as stable (cf. [Desender and Turin, 1989](#_bookmark6)).

We attributed, where possible, ecological and life history traits and conservation priorities to each species: biotope preference (generalists, heathland, nutrient-poor dry biotopes, wet biotopes, woodland – [Desender et al., 2008b](#_bookmark13)), activity period (Spring, Au- tumn – [Desender, 1986; Turin, 2000](#_bookmark6)), distribution range (Atlan- tic-Mediterranean, Atlantic, Central-European, Coastal, European, Middle-European, Northern-European – [Turin, 2000](#_bookmark27)), marginality of the Belgian distribution (i.e., the position of Belgium compared to total distribution area: central, limit – [Desender, 1986](#_bookmark6)), wing form (brachypterous, dimorphic, macropterous – [Turin, 2000; Des-](#_bookmark27) [ender, 1989](#_bookmark27)), size class (Big (>10 mm), small (5–10 mm), very small (<5 mm) – [Desender et al., 2008b](#_bookmark13)) and former threat status in NW Europe (as evaluated by [Desender and Turin (1989)](#_bookmark6): not threatened, threatened, seriously threatened; an overview of the present Red List status of all species according to the IUCN criteria is given in [Desender et al., 2008a](#_bookmark9)). An overview of the traits per species is given in Appendix.

To examine whether the observed changes in distribution area were equally distributed over different classes of ecological and life history traits, we applied a v2-test to compare the frequencies of species in the three trend classes (increasing, stable or decreasing between the different periods). Additionally, we used a Redun- dancy Analysis (RDA) in CANOCO 4.5 for Windows ([ter Braak and](#_bookmark27) [Šmilauer, 2002](#_bookmark27)) to explore how the joint variation of life history traits and threat status were associated with the trends between the different periods. The result is a graphical representation of the distribution of species’ trends with respect to their life history, ecological and biological traits.



Fig. 2. Species richness per grid cell in Belgium in the three investigated periods: a = before 1950; b = 1950–1985; c = 1986–2008; d = grid cells used for analysing the distribution data (i.e., that have at least 25% of their area in Belgium and were well surveyed in the three periods, *N* = 293).

Table 2

Results of the *G*-test comparing the number of observed and expected species that decrease, remain stable or increase per life history or ecological trait. *N* = number of species per trait used in the analysis; 12 = between period 1 and 2, 23 = between period 2 and 3, 13 = between period 1 and 3. +++, highly signiﬁcantly (*p* < 0.001) more, less species than expected decreased (D), remained stable (S) or increased (I) in this trait, ++, very signiﬁcant (*p* < 0.01), +, signiﬁcant (*p* < 0.05), (+), ( ) tendency (*p* < 0.1), ns not signiﬁcant.

|  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | D12 | S12 | I12 | *N* | D23 | S23 | I23 | *N* | D13 | S13 | I13 | *N* |
| *Biotope (b)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Dry biotopes (DB) | +++ |   |   | 99 | ns | ns | ns | 79 | +++ |   |   | 100 |
| Generalists (G) |   | ns | +++ | 78 | ns |   | + | 78 |   | ( ) | +++ | 78 |
| Heathlands (HL) | +++ | ns |   | 27 | ns | ns | ns | 27 | ns | ns | ns | 28 |
| Wet biotopes (WB) |   | +++ | ( ) | 112 | ns | (+) |   | 98 | ns | +++ |   | 113 |
| Woodlands (WL) | ns | ns | ns | 34 | ns | ns | ns | 33 | ns | ns | ns | 36 |
| *Threat status (t)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Not threatened (NT) |   | +++ | +++ | 237 | ns | ns | ns | 225 |   | +++ | +++ | 240 |
| Threatened (T) | +++ |   |   | 75 | ns | ns | ns | 58 | +++ |   |   | 76 |
| Seriously threatened (ST) | +++ |   |   | 42 | ns | ns | ns | 36 | +++ |   |   | 42 |
| *Wing form (w)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Brachypterous (B) | ns |   | ns | 46 | ns | ns | ns | 44 | ns | ns | ns | 46 |
| Dimorphic (D) |   | ns | ++ | 57 | ( ) | ns | ns | 57 |   | ns | +++ | 57 |
| Macropterous (M) | (+) | ns |   | 238 | ns | ns | ns | 212 | +++ | ns |   | 238 |
| *Activity period (a)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Spring (S) | ns | ns | ns | 266 | ns | ns | ns | 243 | ns | ns | ns | 268 |
| Autumn/summer (A) | ns | ns | ns | 84 | ns | ns | ns | 75 | ns | ns | ns | 85 |
| *Range (r)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Atlantic-Mediterranean (AM) | ns | ns | ns | 17 | ns | ns | ns | 15 | ns | ns | ns | 18 |
| Atlantic (At) | ns | ns | ns | 11 | ns | ns | ns | 10 | ns | ns | ns | 11 |
| Central-European (CE) | ns | ns | ns | 90 | ns | ns | ns | 81 | ns | ns | ns | 90 |
| Coastal (Co) | ns | (+) | ns | 17 | ( ) | ns | ns | 13 | ns | ns | ns | 18 |
| European (Eu) |   | ns | + | 42 | ns |   | ns | 43 |   | ns | + | 43 |
| Middle-European (ME) | ns | ns | ns | 159 | ns | ns | ( ) | 144 | ns | ns | ns | 157 |
| Northern-European (NE) | ns | (+) | ( ) | 13 | ns | ns | ns | 11 | ns | ns | ns | 14 |
| *Marginality (m)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Central (C) | ns | ns | +++ | 246 | ns | ns | ns | 234 | ns | ns | (+) | 248 |
| Limit (L) RarityP1 (p1) | ns | ns |   | 106 | ns | ns | ns | 84 | ns | ns | ( ) | 108 |
| Very rare (VR) | ns | + | ns | 45 |   |   |   |   | ns | ns | ns | 49 |
| Rare (R) | +++ | ns |   | 193 |   |   |   |   | ++ | ns |   | 193 |
| Common (C) | ns | ns | +++ | 82 |   |   |   |   |   | ns | +++ | 82 |
| Very common (VC) |   | ns | +++ | 34 |   |   |   |   | ( ) | ns | ++ | 34 |
| *RarityP2 (p2)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Very rare (VR) |   |   |   |   | ++ | +++ | ns | 93 |   |   |   |   |
| Rare (R) |   |   |   |   | (+) | ( ) | ns | 148 |   |   |   |   |
| Common (C) |   |   |   |   | ns | ns | + | 53 |   |   |   |   |
| Very common (VC) |   |   |   |   | + | ns |   | 25 |   |   |   |   |
| *Size (s)* |  |  |  |  |  |  |  |  |  |  |  |  |
| Big (B) | ++ |   | ns | 78 | + | ns | ns | 64 | +++ |   | ns | 79 |
| Small (S) | ns | ns | ns | 168 | ns | ns | ns | 148 | ns | ns | ns | 168 |
| Very small (VS) |   | +++ | ns | 108 | ns | ns | ns | 107 |   | ++ | ns | 111 |
| *Genus (g)* |  |  |  |  |  |  |  |  |  |  |  |  |
| *Agonum* (Ag) |   | ns | + | 21 | ns | ns | ns | 20 |   | ns | + | 21 |
| *Amara* (Am) | + | ( ) | ns | 34 | ns | ns | ns | 30 | ns | ns | ns | 34 |
| *Bembidion* (Be) |   | +++ | ns | 56 | ns | ns | ( ) | 53 | ns | + |   | 58 |
| *Carabus* (Ca) | ns | ns | ns | 14 | ++ | ns | ( ) | 14 | ns | ns | ns | 14 |

*Dyschirius* (Dy) ns (+) ns 11 ( ) ns ns 8 ns ns ns 11

*Harpalus* (Ha) +++ ( ) 39 ns ns ns 30 ++ ns 39

*Pterostichus* (Pt) ns ns ns 24 ( ) ns +++ 21 ( ) ns +++ 24

1. Results

Out of the 397 species present in Belgium during the period

<1950, 46 carabid beetle species were no longer recorded during the period 1950–1985. Two species were added to the Belgian fau- na in the period 1950–1985, resulting in a net loss of 44 species. Twelve species that were present in the ﬁrst, but that were not re- corded in the second period, were rediscovered in the period 1986–2008, indicating that collecting efﬁcacy might have in- creased in the third period. Five more new species were added to the Belgian fauna since 1986. Comparing the period 1986–2008 and the period <1950, 10 additional species were no longer re- corded. This resulted in a net loss of 37 species compared to the period <1950 (44 species no longer recorded and seven new spe- cies – [Table 1](#_bookmark1)).

From the 149 species that declined from the period <1950 to the period 1950–1985, 27 species continued to decline in the per- iod 1986–2008, 60 species remained stable and 30 species in- creased (a further 32 species became too rare in the two recent periods and could not be analysed – Appendix). However, from the 149 species that declined between the ﬁrst two periods, 116 species (78%) did not regain their historical distribution range in the period 1986–2008 ([Table 3](#_bookmark4)). On the other hand, from the 68 species that increased between the ﬁrst two periods, 48 species continued to increase or remained stable and 20 species de- creased between the period 1950–1985 and the period 1986– 2008. From the 68 species that increased between the ﬁrst two periods, 49 species increased even more in distribution area com- pared to their historical distribution, 17 species fell back to their historical distribution range in the third period and two species

Table 3

Results of the *G*-test comparing the number of observed and expected species that decrease, remain stable or increase among periods. +++, highly signiﬁcantly (*p* < 0.001) more, less species than expected decreased (D), remained stable (S) or increased (I), ++, very signiﬁcant (*p* < 0.01), +, signiﬁcant (*p* < 0.05), (+), ( ) tendency (*p* < 0.1), ns not signiﬁcant. *N* = number of species used in the analysis; 12 = between period 1 and 2, 23 = between period 2 and 3, 13 = between period 1 and

3.

|  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Trend | D23 | S23 | I23 | *N* | D13 | S13 | I13 | *N* |
| D12 | ns | ns | (+) | 149 | +++ |   |   | 165 |
| S12 | ns | ++ |   | 135 |   | +++ | ( ) | 115 |
| I12 | ns |   | ns | 68 |   | ns | +++ | 72 |
| D13 | +++ | ns |   | 73 |   |   |   |   |
| S13 |   | + | ns | 177 |   |   |   |   |
| I13 |   |   | +++ | 67 |   |   |   |   |

declined compared to their historical distribution ([Table 3](#_bookmark4), Appendix).

Between the period <1950 and the period 1950–1985, the num- ber of decreasing species was especially high among the species from nutrient-poor dry biotopes and heathlands (e.g., *Acupalus brunnipes*, *Amara equestris*, *Bembidion humerale*, *Calathus cinctus*), threatened species, macropterous species, rare species and big spe- cies (e.g., *Amara* spp. and *Harpalus* spp. – [Table 2](#_bookmark3)). More species than expected increased in distribution area in the following eco- logical categories: generalists, non-threatened species, dimorphic species, species with a pan-European distribution range, species in the centre of their distribution range and common species (e.g., *Agonum* spp. – [Table 2](#_bookmark3)). The Redundancy Analysis (RDA) addi- tionally showed that a relatively large number of very small spe- cies and species from wet biotopes remained stable (e.g., *Bembidion* spp. and *Dyschirius* spp. – [Fig. 3](#_bookmark5)a).

Between the period 1950–1985 and the period 1986–2008, more rare and very common species and big species decreased than expected, while fewer than expected dimorphic species de- creased (e.g., *Carabus* spp.). Species that increased more than ex- pected were rather numerous among the generalists and common species (e.g., *Pterostichus* spp. – [Table 2](#_bookmark3)). The Redundancy Analysis (RDA) additionally showed that a rather large number of rare species, coastal species, species from wet biotopes and mac- ropterous species remained stable (e.g., *Dyschirius* spp.) and that a large number of dimorphic species increased their range size (e.g., *Pterostichus* spp. – [Fig. 3](#_bookmark5)b).

Comparing the period 1986–2008 with the period <1950 largely conﬁrmed the trends between the ﬁrst two periods ([Table 2](#_bookmark3)). The Redundancy Analysis (RDA) additionally showed that more species from nutrient-poor dry biotopes, rare species, macropterous spe- cies and big species decreased in distribution area (e.g., *Harpalus* spp.), whereas many species from wet biotopes (e.g., *Bembidion normannum*, *Parophonus maculicornis*) and very small species re- mained stable (e.g., *Dyschirius* spp.), and that a large number of generalists, dimorphic species and species with a European distri- bution increased in range size (e.g., *Agonum* spp. – [Fig. 3](#_bookmark5)c).

1. Discussion

Twenty years after a ﬁrst analysis ([Desender and Turin, 1989](#_bookmark6)), we re-examined the changes in species composition of carabid beetles in Belgium to detect whether the previously detected trends have been altered during the last decades. Compared to the ﬁrst half of the 20th century, 44 species were no longer re- corded, while seven species were added to the Belgian fauna. By relating the changes in distribution area to species’ ecological and life history traits, we were able to identify ecological trait assemblages showing the largest changes. Large carabids, species from nutrient-poor dry biotopes and species that were already

threatened, continued to decline since 1986. On the other hand, generalists, species from wet biotopes, non-threatened species, common species and species that occur in the centre of their range showed an increasing or a stable trend in distribution area. Some species that declined between the ﬁrst two periods showed an increasing or stable trend between the last two periods. The major- ity of these species, however, did not regain their historical distri- bution area in the most recent period.

* 1. *Data quality*

Due to differences in collection methods and the length of the time periods considered ([Fig. 1](#_bookmark0)), comparisons of former and pres- ent-day distribution data, are often biased in time and in space (e.g., [Dennis et al., 1999; Dennis and Thomas, 2000](#_bookmark6)). Furthermore, collecting in the past often focused on big species and the more common species were often under-collected. Since carabid beetles are collected in a standardised way by means of pitfall traps and usually have to be classiﬁed using a stereomicroscope, this possible bias in collecting effort is nowadays strongly reduced. Pitfall trap- ping is a non-selective sampling method, with which both rare and common species are collected. By reducing the analysis to a set of squares that were at least minimally surveyed in all three periods, we additionally lowered a possible spatial bias (cf. [Maes and Van](#_bookmark28) [Dyck, 2001](#_bookmark28)). The ﬁnal set of squares and species used in the anal- ysis is believed to be representative for carabid diversity in Bel- gium. Most of the Belgian ecological regions ([Dufrêne and](#_bookmark17) [Legendre, 1991](#_bookmark17)) are represented with at least 20% of their grid cells in the analysis ensuring a good coverage of the data over the coun- try. Although we used arbitrary criteria to determine which species and squares were sufﬁciently surveyed in all periods, other, more strict criteria yielded very similar results. On the basis of grid cells, however, most of the common, generalist species seem to have sta- ble or even increasing distributions, but it is fairly difﬁcult to de- tect trends among periods on a population level (i.e., the number of individuals). For butterﬂies, it has been shown that relatively widespread species can show a stable or increasing trend in distri- bution, but a relatively strong decrease in abundance ([Van Dyck](#_bookmark27) [et al., 2009](#_bookmark27)). Pitfall trapping only allows for such comparisons if the dataset is very large, the sampled region is large enough and the sampling period is long. In Belgium, such a data set is only available in the dune area ([Desender, 1996](#_bookmark6)).

Some of the grid cells used in the analysis are situated near big cities such as Brussels, Antwerp, Gent, Liege and Namur ([Fig. 2](#_bookmark2)d). The observed trends between periods are therefore probably a combination of both increasing urbanization in the neighbourhood of these cities (leading to the destruction of biotopes – [Gaublomme](#_bookmark21) [et al., 2008](#_bookmark21)) and a more general degradation of habitat quality in grid cells that are situated in the remaining semi-natural areas such as heathlands in the northeast, calcareous grasslands in the south and dune grasslands along the coast ([Niemelä et al., 2002](#_bookmark35)).

Surprisingly, more species than expected that decreased in dis- tribution area from the period <1950 to the period 1950–1985, in- creased in the period 1986–2008. Many of these species were rare in the ﬁrst period and were restricted to semi-natural biotopes such as nutrient-poor dry grasslands, heathlands or wet biotopes. The threat status of many of these species ([Desender et al., 1995](#_bookmark7)) may have caused a bias in the sampling effort in their threatened biotopes (e.g., heathlands ([Versteirt et al., 2002](#_bookmark36)), dry or wet nutri- ent-poor grasslands, river banks) ([Lambeets et al., 2008, 2009](#_bookmark28)).

* 1. *Ecological and life history traits*

The strongest declines in distribution area compared to the per- iod <1950 (which can be seen as a ‘‘benchmark” period) were found in species from nutrient-poor dry biotopes, threatened



Fig. 3. Redundancy Analysis plot of the trends between the different periods (a = period12, b = period 23, c = period 13) as ‘‘environmental” variables (black dots), the genuses (diamonds) and the threat status (open circles) as supplementary variables and the life history and ecological traits as active variables (arrows). The abbreviations used in the graph can be found in [Table 2](#_bookmark3). Only the environmental variables that are most correlated with the ﬁrst two axes are shown.

species, rare species, macropterous species and big species. On the other hand, generalists, non-threatened species, dimorphic species, species with a pan-European distribution range, species in the cen- tre of their distribution consistently increased compared to this benchmark period. Species from wet biotopes and very small spe- cies, in general, remained stable.

Many semi-natural biotopes (e.g., dry and wet heathlands, nutrient-poor dry or calcareous grasslands, dunes, woodlands) have decreased strongly in area in Belgium leading to a very strong

decline in biodiversity (e.g., [De Bruyn et al., 2003; Maes and Van](#_bookmark22) [Dyck, 2001](#_bookmark22)). Additionally, habitat quality within these remnants has declined simultaneously due to an intensiﬁcation of agricul- tural practices and to a very high level of nitrogen deposition ([Schneiders et al., 2007](#_bookmark27)). In Belgium, but also in other strongly industrialised NW European regions such as The Netherlands or N-France, specialist carabid beetles usually occur in small and highly fragmented semi-natural remnants ([Desender and Turin,](#_bookmark6) [1989; Desender et al., 1994b; Davies and Margules, 1998;](#_bookmark6)

[Hendrickx et al., 2009](#_bookmark6)). This makes the extinction risk for rare and specialist carabid beetles much higher than for generalists that are able to tolerate much higher levels of disturbance and occur in more widespread biotopes (e.g., intensively used pastures and ara- ble land – [Niemelä, 2001a; Kotze et al., 2003](#_bookmark32)). Species from wet biotopes seem to be less affected by the loss of biotope quantity and/or quality (almost 50% of them remained stable or increased compared to the ﬁrst half of the 20th century). One of the impor- tant reasons is probably the improved water quality in Belgium in the second half of the last century ([Schneiders et al., 2007](#_bookmark27)).

Over all periods, more large species than expected declined, which is consistent with previous analyses ([Kotze and O’Hara,](#_bookmark25) [2003](#_bookmark25)). Large species usually have smaller populations and lower reproduction rates and, therefore, respond more slowly to environ- mental changes ([Simberloff, 1994; Beissinger, 2000; Davies et al.,](#_bookmark27) [2004; Gibb et al., 2006](#_bookmark27)). In Belgium, most of the big carabid species are brachypterous which makes them additionally vulnerable for habitat fragmentation.

A more surprising result is that more dimorphic species than expected increased compared to the period <1950. Dimorphic spe- cies (i.e., a part of the population is capable of ﬂight while another part has no wings at all) are able to invest maximally in reproduc- tion in stable environmental situations but also to leave sites that have become unsuitable and colonize new sites ([Kotze and O’Hara,](#_bookmark25) [2003](#_bookmark25)). Once a new site has been colonized, the proportion of winged individuals will decrease again (e.g., [den Boer, 1970; Des-](#_bookmark26) [ender et al., 1998](#_bookmark26)).

* 1. *Carabid beetles and conservation*

Carabid beetles were one of the ﬁrst invertebrates for which a Red List was compiled in Belgium ([Desender et al., 1995](#_bookmark7)). Conser- vation policies or management measures are often based on ‘clas- sical’ species groups such as birds, mammals and/or plants ([Maes](#_bookmark28) [et al., 2005](#_bookmark28)). Invertebrates in general, and carabid beetles in partic- ular, however, can be complementary to these groups because they are very species-rich, abundant and generally operate on a smaller spatial scale. Furthermore, they occur in almost all biotope types including some that are not inhabited by many other species such as salt marshes, river banks and/or bare sandy plains.

The decline of carabid beetles in Belgium between the period

<1950 and the period 1950–1985, has been halted for a consider- able number of species. In the period 1986–2008, however, 60% of these species still had not reached the same distribution area as in the ﬁrst half of the 20th century. Most of these species now only occur in large and high-quality nature reserves with the last remnants of semi-natural biotopes and have, at present, little or no possibilities to further increase their distribution range. The conservation of carabid beetles in Belgium should, therefore, be undertaken on two different levels: (1) improving or restoring the quality and quantity of preferably large threatened (semi-)nat- ural habitat sites, and (2) improving connectivity among sites to make exchange and/or colonization more easy for species with low dispersal capacities.

Restoration and management of threatened (semi-)natural bio- topes should focus on those biotopes in which a large number of species continued to decline, i.e., nutrient-poor dry grasslands, chalk grasslands ([WallisDeVries et al., 2002](#_bookmark39)) and heathlands ([Gard-](#_bookmark20) [ner, 1991](#_bookmark20)). Since large sites have more species, we should aim at the conservation and management of large natural sites as core areas for carabid beetle diversity ([den Boer, 1970; de Vries,](#_bookmark26) [1994](#_bookmark26)). Management measures here could consist of a better pro- tection status of sites, with an appropriate nature management taking the requirements of the most threatened carabid beetles and other insects into account. Many abandoned grasslands and heathlands in NW Europe are now overgrown (by bushes and *Moli-*

*nia caerulea* respectively), which substantially cools down the microclimate at ground level ([WallisDeVries and van Swaay,](#_bookmark37) [2006](#_bookmark37)). Specialist carabid beetles of dry nutrient-poor biotope and heathland need a warm microclimate that can be created or re- stored by mowing or relatively intensive grazing regimes. To en- large the area of nutrient-poor dry biotopes, set-aside ﬁelds or former arable sites can be converted to nutrient-poor grassland or heathlands for threatened carabid beetles (e.g., *Amara tricuspi- data*, *Harpalus froelichi* or *Harpalus griseus* – [Desender and Bos-](#_bookmark6) [mans, 1998; Versteirt et al., 2002](#_bookmark6)). Such sites can be colonized more easily if they are in the immediate vicinity of suitable grass- lands and/or heathlands. However, the conservation of nutrient- poor biotopes in Belgium and other NW European countries or re- gions is seriously hampered by the very high nutrient deposition values (on average 33 kg N/ha/year with peaks up to 72 kg N/ha/ year – [Schneiders et al., 2007](#_bookmark27)). Removing the nutrient-rich top layer of the soil usually cannot compensate for the high amount of nitrogen deposition. Therefore, a more general, region-wide, reduction of fertilizer use in agriculture should accompany restora- tion measures. Although the ecological group of forest inhabiting carabid beetles did not show a signiﬁcant decline, some of the typ- ical woodland species are severely threatened in Belgium ([Desend-](#_bookmark9) [er et al., 2008a](#_bookmark9)). For typical woodland species it is, therefore, important to maintain large and uneven-aged forests with small clear-cuttings and indigenous tree species ([du Bus de Warnaffe](#_bookmark23) [and Lebrun, 2004](#_bookmark23)). Apart from conserving large (semi-)natural areas, the conservation of rare and small biotopes is also important for some small relic populations of threatened species because they often contain unique genotypes that are absent from large similar biotope types: salt marshes (e.g., *Dicheirotrichus gustavii* and *Pogonus chalceus* – [Desender et al., 1998, 2007](#_bookmark8)), tidal marshes (e.g., *Bembidion normannum* and *Bembidion maritimum* – [Desender](#_bookmark6) [and Maelfait, 1999](#_bookmark6)), stony and sandy river banks (e.g., *Bembidion atrocoeruleum*, *Bembidion punctulatum*, *Tachys micros* and *Tachys parvulus* – [Lambeets et al., 2008, 2009](#_bookmark28)).

In highly fragmented regions such as NW Europe, a second important conservation measure to preserve to most threatened carabid beetles is to restore or create a higher degree of connectiv- ity among core semi-natural areas. For example, many of the spe- cialist woodland carabid beetles (e.g., *Carabus* spp.) are brachypterous and highly affected by habitat fragmentation ([Magura et al., 2001; Niemelä, 2001a](#_bookmark28)). Studies have shown that hedgerows connecting forest patches are indeed used by forest carabids ([Charrier et al., 1997; Major et al., 1999; Niemelä,](#_bookmark18) [2001b](#_bookmark18)), but most probably only by the relatively generalist forest species ([Koivula and Vermeulen, 2005](#_bookmark24)). For the specialist forest species, only an appropriate management scheme and the enlarge- ment of forests is, therefore, helpful for their conservation. Road verges as corridors between dry nutrient-poor biotopes only ap- pear to work on relatively short distances (e.g., 50–150 m/year for heathland species) and often act more as sink biotopes than as corridors ([Vermeulen, 1994; Eversham and Telfer, 1994](#_bookmark33)). Efforts to conserve species from dry nutrient-poor biotopes and heath- lands should, therefore, focus more on the conservation and resto- ration of suitable sites than on the creation of corridors.

The use of ecological and/or life history traits permits to detect patterns across taxonomic groups (e.g., birds – [Okes et al., 2008](#_bookmark38), carabid beetles and spiders – [Lambeets et al., 2009](#_bookmark28), moths – [Mat-](#_bookmark31) [tila et al., 2009](#_bookmark31)) and makes it possible to draw more general con- clusions on what type of species are expanding or declining. This allows conservation biologists to use species from less well-stud- ied taxonomic groups with similar ecological and/or life history traits as those of relatively well-studied decreasing species in pol- icy making. Furthermore, the use of ecological and/or life history traits facilitates the communication with policy makers because they do not have to know the detailed ecology and biology of the

species as such. A description of the type of species (e.g., biotope type, dispersal abilities, size) that are in need of the most urgent conservation is much more understandable than long (Red) lists of scientiﬁc species names.

Species groups with a relatively long tradition of collecting dis- tribution data are ideal indicators to detect whether objectives such as halting or reducing the loss of biodiversity have been achieved (Countdown2010 – [Mace and Baillie, 2007](#_bookmark28)). For a small number of groups (e.g., birds, plants, mammals, butterﬂies) such relatively long-term data are readily available now on country, European or even global levels. In many countries, carabid beetles have been sampled for a long time and the information from this species-rich and easy-to-sample invertebrate group, would cer- tainly be complementary to that of other species groups ([Maes](#_bookmark28)  [and Van Dyck, 2005](#_bookmark28)). In future, a monitoring scheme using an indi- cator to detect whether the loss of biodiversity has been stopped, would be recommendable. Such a composed indicator could con- sist of a set of complementary species from different taxonomic groups for which enough information is available on both ecologi- cal and/or life history traits and conservation priorities. This type of indicator would be understandable for policy makers and for the  wider audience and would greatly facilitate the communication about biodiversity conservation ([Pullin, 2002](#_bookmark40)).

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at [doi:10.1016/j.biocon.2010.03.039](http://dx.doi.org/10.1016/j.biocon.2010.03.039).

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