The use of indicator species in nature management and policy making

The case of invertebrates in Flanders [northern Belgium]

Dirk Maes

Universiteit Gent

Faculteit Wetenschappen

Academiejaar 2003-2004

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Het gebruik van indicatorsoorten in het natuurbeheer en -beleid

Ongewervelden in Vlaanderen als voorbeeld

door

Dirk Maes

Thesis submitted in fulfilment of the requirements for the degree of Doctor [Ph.D.] in Sciences

Proefschrift voorgedragen tot het bekomen van de graad van Doctor in de Wetenschappen

Promotor: Prof. Dr. Eckhart Kuijken [Vakgroep Biologie – Universiteit Gent]

Co-promotor: Dr. Hans Van Dyck [Departement Biologie – Universiteit Antwerpen]

"As the laws of Nature must be the same for all beings, the conclusions furnished by this group of insects must be applicable to the whole organic world; therefore the study of butterflies – creatures selected as the types of airiness and frivolity – instead of being despised, will some day be valued as one of the most important branches of biological sciences."

Henry Walter Bates [1864]. Naturalist on the Amazone. J. Murray, London.

Acknowledgements

In de eerste plaats wil ik de beide promotoren, Eckhart en Hans, bijzonder hartelijk bedanken voor het vertrouwen en voor de aangename samenwerking. Eckhart gaf me steeds de nodige vrijheid bij de wetenschappelijke invulling van mijn taken op het Instituut voor Natuurbehoud en Hans was [en is dat nog steeds] de immer kritische, maar constructieve wetenschapper waarmee het bijzonder prettig samenwerken was, en zeker zal blijven! Een meer complementair promotorenduo is moeilijk voor te stellen ...

Dirk B. ben ik bijzonder dankbaar omdat hij mij de eerste stappen leerde zetten in het schrijven van wetenschappelijke artikels, maar ook omdat hij altijd tijd maakte om constructieve kritiek te leveren en oplossingen te vinden voor allerlei statistiekproblemen. Merci Dirk!

All co-authors of the different chapters are greatly acknowledged: Hans Van Dyck [UA], Wouter Vanreusel [UA], Marius Gilbert [ULB], Anny Anselin [IN], Dirk Bauwens [IN], Joeri Cortens [UA], Luc De Bruyn [IN], Geert De Knijf [IN], Roger Dennis [Oxford Brookes University], Philippe Goffart [OFFH], Willem Talloen [UA], Nicolas Titeux [UCL] and Chris van Swaay [De Nederlandse Vlinderstichting]. I also kindly thank the other members of the jury for their constructive remarks on the manuscript version of this thesis: Andrew Pullin [School of Biosciences, Birmingham], Marc Dufrêne [DGRNE-OFFH, Gembloux], Steven Degraer [UGent], Maurice Hoffmann [UGent], Luc Lens [UGent], Jean-Pierre Maelfait [UGent] and Magda Vincx [UGent].

Het uitgebreide A-team [waarbij 'A' staat voor *alcon*, met Hans Van Dyck, Wouter Vanreusel, Joeri Cortens, Sofie Regniers en Raf Baeyens] maakten het vele mieren- en gentiaanblauwtjesveldwerk tot een bijzonder aangename periode. Andere veldwerkers, determinatoren of gegevensaanleveraars ben ik eveneens dankbaar: Leon Baert, Yvan Barbier, Jean-Yves Baugnée, Inge Brichau, Benoit De Bast, Wouter Dekoninck, Konjev Desender, Graham Elmes, Violaine Fichefet [un grand merci pour la traduction du résumé!], Patrick Grootaert, Jos Gysels, Frederik Hendrickx, Maarten Jacobs, Koen Lock, Jean-Pierre Maelfait, Hans Matheve, Marc Pollet, het KMI, François Vankerkhoven, Lucien Verlinden, Tom Verschraegen en Charles Verstraeten. Dank ook aan de conservators en natuur- en boswachters die 'hun' reservaten openstelden voor wetenschappelijk onderzoek: Joost Dewyspelaere, Jan Dirckx, Eddy Feyen, Dirk Geysels, Jos Gorissen, Gerard Jannis, Freddy Janssens, Jos Jaspers, Ignace Ledegen, Jef Leestmans, Hubert Lehaen, Harry Lesseliers, Gie Luyts, Karel Molenberghs, Ghis Palmans, Willy Pardon, Alex Riemis, Marc Schuermans, Marc Smets, Willy Vanlook, Marcel Van Waerebeke, Tom Verschraegen en Manu Vlaeyens. Bijzonder veel dank gaat uit naar de honderden vrijwilligers van de verschillende inventarisatieprojecten die de hier voorgestelde analyses mee mogelijk gemaakt hebben.

De collega's op 'het instituut' verdienen een welgemeend woord van dank: zij maken het IN tot een bijzonder aangename werkplek waar je met evenveel enthousiasme je mening kwijt kan over natuurbehoud als over zovele andere 'dingen des levens'. Een welgemeende dankuwel aan mijn spitsbroeder Jan voor de interessante en fijne babbels tijdens de laatste loodjes van dit doctoraat.

Un grand merci aussi aux Amefessiens [l'équipe la plus intelligente du monde?] de pouvoir dewanner et boire des chopes chez les tapettes tous les mardis ...

Mijn ouders hadden altijd graag een 'dokter' in de familie gehad, maar ze zullen het met een 'doctor' moeten stellen. Zonder hun enorme dosis vertrouwen tijdens mijn biologiestudies, was er nooit een doctoraat gekomen. Dankuwel!

Ariane, Margot en Oscar hebben het mogelijk gemaakt om een gezin te combineren met intensieve veldwerk- en schrijfperiodes. Hun bijdrage aan dit 'boekje' is dan ook veel groter dan ik in woorden kan uitdrukken. Nu het 'eitje' gelegd is, zal er weer wat meer tijd zijn voor de vlinders in de buik ...

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1 Introduction, aims and outline

FOTO: DIRK MAES

"There is an implicit tendency in much of the thinking in conservation biology to assume that species are the vulnerable entities, and that ecosystems are somehow more permanent. ... In reality, it is species that are the constant elements and ecosystems that are transitory. In the very long term, it follows that species conservation, and the vagaries of luck and politics will determine what kind of ecosystems might exist, because ecosystems are more ephemeral than species. We must therefore do what we can now to preserve both species and ecosystems; ecosystems because species need them in the short-term, and species because they make ecosystems in the long term."

John Lawton [1997]. The science and non-science of conservation biology. *Oikos* 97: 3-5.

Introduction

As in most other western countries with a high pressure on the open space and thus on biodiversity, decisions in nature conservation are often non-ecologically based [e.g., political agreements with other land-users, socio-economic priorities], or in the best case, ecologically based within very strict boundaries of non-ecological arguments. Nature conservation in Flanders, but also elsewhere, is primarily site-based rather than species-based [Franklin 1993]. This means that the acquisition and management of sites is mainly based on the presence of biotopes [e.g., EU Habitat Directive] or the [assumed] maintenance of ecological processes [e.g., nutrient cycles, hydrology], but not necessarily on the occurrence or state of particular species. The Flemish Ecological Network is one example of this site-focused policy based on the presence of biotopes [De Blust & Kuijken 1996; Kuijken & De Blust 1997; Vanholen *et al.* 2003]. Such approaches can, however, lead to the undetected loss of species [e.g., Pickett *et al.* 1992], since particular [micro-]habitats of a number of, especially invertebrate, species [Thomas 1994] can rapidly change [vegetation height, presence of host plant species, etc.] without an obvious or detectable change of the overall ecological processes [Kareiva & Levin 2003]. Furthermore, the effectiveness of such holistic approaches is difficult to evaluate because the objectives are often only vaguely described [Simberloff 1998]. In order to evaluate the effectiveness of these, mainly site-based, decisions, clear objectives and assessment criteria should be formulated [Caro & O'Doherty 1999; Hilty & Merenlender 2000; Noss 1990].

The ecological information content of species can have a considerable contribution to nature conservation applications. However, knowledge about minimum habitat size and habitat quality requirements and other autecological information [e.g., relations with other species, trophic level, mobility, etc.] of the species has to be available in order to apply species information for nature conservation purposes [Poiani *et al.* 2000; Root *et al.* 2003;

Wallis de Vries 1999]. This limits the number of possible species considerably. Although such information is available for a variety of taxonomic groups, the use of species in nature conservation in a well-organised, scientific way is still in its infancy in Flanders [Van Dyck *et al.* 1999]. Due to the long list of species on the EU Bird Directive and the availability of distribution information for many of the listed species, birds are one of the scarce examples of species used for delineating conservation areas [Bird directive areas or Ramsar] in Flanders.

The use of species in nature conservation

Species can play a significant role in nature conservation, either as goals [target species] or as tools [indicator species]. When species are used as goals, success can be measured by an increase in population numbers, the number of recolonizations or an expansion of the species' distribution area, etc. But, species have the benefit that several requirements relating to habitat quality, quantity and geometry can be defined or at least estimated. This ecological knowledge may consequently be used as a tool in management planning and/or evaluation, site selection, etc. [Hilty & Merenlender 2000; Mc Geoch 1998]. But then the question arises, which species to work with? Available knowledge is an obvious bottleneck. Moreover, conservation practitioners request, preferentially, rather simple, straightforward approaches that can be readily implemented by non-experts in order to keep efforts within limited time and financial budgets [Fleishman *et al.* 2000].

The use of short-cut concepts like indicator species to protect, manage or restore habitats and local biodiversity is very appealing, but poses some practical problems. A variety of terms concerning indicators are used in both scientific literature and in policy making documents which hampers a clear communication about the use of indicator species in nature conservation [Van Dyck *et al.* 2001]. With respect to the use of species in conservation, three basic concepts can be recognized: flagship species, target species and indicator species. *Flagship species* serve to increase the awareness of conservation needs by helping to gain public and political sympathy based on their appeal to people such as the Giant Panda [WWF], whales [Greenpeace], etc. [Landres *et al.* 1988; New 1995c; Simberloff 1998]. *Target species* are used as goals, i.e., the conservation of the species *per se* is the goal [Simberloff 1998]. This typically regards locally or more widely threatened or rare species, although there is a bias to consider flagship species as goals as well. Examples of target species can be found in annex II of the EU Habitat and Bird Directives. A considerable consent exists about the use of the terms flagship species and target species. Far more problematic is the use of the term *indicator species* because different authors apply different definitions and interpretations of an indicator [Landres *et al.* 1988; Simberloff 1998]. Indicator species indicate a particular suite of environmental conditions or the state/health of other sympatric species [Dale & Beyeler 2001; Landres *et al.* 1988]. A particular type of indicator species is the so-called umbrella species, i.e., its conservation serves to protect sympatric species [Fleishman *et al.* 2001b; Launer & Murphy 1994]; examples of umbrella species are usually large mammals, like rhinos or cougars [e.g., Beier 1993; Wilcox 1984], birds [e.g., Martikainen *et al.* 1998; Mikusinski *et al.* 2001; Rubinoff 2001] or vascular plants [e.g., Oliver *et al.* 1998; Pharo *et al.* 1999]. Some of these organisms have the additional benefit of being flagship species [Dietz *et al.* 1994; Landres *et al.* 1988; New 1995c; Simberloff 1998]; among invertebrates, the Bay checkerspot butterfly [*Euphydryas editha bayensis*] has been identified as umbrella species [Launer & Murphy 1994].

Another problem with the use of indicator species is that the effectiveness of the indicator concept is often assumed, but is rarely underpinned by sound scientific research [Andelman & Fagan 2000; Andersen 1999; Fleishman *et al.* 2001b; Simberloff 1998]. Furthermore, the use of a single species or a single taxonomic group as a conservation umbrella for other sympatric species has been criticized: a single indicator species or group is unlikely to encompass all or most of the ecological requirements [habitat size, habitat configuration, relationships with other species, resources, etc.] of a series of other species, or most of the environmental conditions in a certain biotope or landscape [e.g., Landres *et al.* 1988; Niemi *et al.* 1997; Prendergast *et al.* 1993a]. In contrast with earlier assumptions, species distribution patterns, including diversity hotspots [Myers *et al.* 2000], may differ considerably among taxonomic groups [e.g., Prendergast *et al.* 1993a; van Jaarsveld *et al.* 1998]. Apart from the concept, the definition of umbrella species – its minimum area requirement is at least as comprehensive as the rest of the community [Wilcox 1984] – has shifted to a more area-independent definition: its conservation serves to protect sympatric species [cf. Fleishman *et al.* 2001b; Launer & Murphy 1994].

Recently, several authors have advocated a multi-species approach in conservation biology, i.e., using a group of species instead of a single umbrella species [e.g., Fleishman *et al.* 2001b; Hilty & Merenlender 2000; Lambeck 1997; Root *et al.* 2003]. The underlying rationale is that a carefully selected group of taxonomically and ecologically different species [the so-called multi-taxa or 'shopping basket' approach; Pullin 2002a] is more likely to provide a complementary, integrative picture of the features [and hence the quality or carrying capacities] of a conservation area [or a habitat network] than a single species.

Furthermore, Collins & Thomas [1991] and Samways [1993] among others, have pleaded for a more prominent use of insects and other invertebrates in conservation biology than is currently the case. This may particularly be appropriate in traditionally managed, man-made habitats where habitat specialist insects heavily depend on vegetation structures and associated microclimates that are less relevant to birds or mammals [Murphy & Wilcox 1986; Thomas 1994]. Hence, large species groups like insects and/or other invertebrates should not *a priori* be excluded from such multi-species groups in order to involve ecological aspects at intermediate or even small spatial scales [Brown 1997; Kremen *et al.* 1993; Kotze & Samways 1999; Mc Geoch 1998].

Invertebrates in conservation biology

Conservation biology is a relatively new multi-disciplinary science that has developed in response to the late 20th century extinction crisis [Soulé 1985; Soulé & Wilcox 1980]. It has two major goals: [1] investigate human impact on species, populations and biotopes and [2] develop practical approaches to ensure the conservation of species and ecosystems [New 1995c; Primack 1998]. Conservation biology arose because none of the traditional applied disciplines [e.g., agriculture, wildlife management, forestry, fisheries biology, but also political and social sciences] by themselves were comprehensive enough to address the threats to biodiversity. Another difference between conservation biology and the 'traditional' academic disciplines is that conservation biology tries to provide solutions that can be applied in real world field situations. Conservation biology is a crisis discipline and decisions are often being made under severe time pressures [Pullin & Knight 2001]. Therefore, conservation biologists must be willing to make crucial decisions before they are confident in the sufficiency of the data [Soulé 1985].

Despite their large numbers and their omnipresence, invertebrates have received far less attention in conservation biology than large mammals, birds and vascular plants [New 1995c]. In the early years of conservation biology, the delineation of nature reserves [and other protected areas] and the planning of nature management measures were almost exclusively vertebrate-based. It was assumed that large nature reserves delineated for birds and mammals would maintain invertebrates as well. A further assumption was that a large variety of vascular plants would assure a rich invertebrate fauna [Landres *et al.* 1988; New *et al.* 1995; Simberloff 1998]. It has now become clear that these assumptions were wrong and that, instead, stronger declines and more numerous extinctions were observed in invertebrates compared to vertebrate species [some examples in Flanders are butterflies – Maes & Van Dyck 2001; dragonflies – De Knijf & Anselin 1996; grasshoppers and crickets – Decleer *et al.* 2000]. Why do invertebrates pose different conservation problems than vertebrates [like birds for example]? Three major differences with vertebrate management practices probably caused these stronger declines [Thomas 1994]: [1] many invertebrates occupy very narrow niches [habitats] within one or more biotopes, [2] invertebrates can sometimes persist on very small habitat patches [< 1 ha] that only remain suitable for a short time period [3-10 years] and [3] at least several invertebrates are too sedentary to colonize new patches even within relative short distances [300 m – 1 km] which makes them more sensitive to fragmentation than other more mobile taxa [e.g., Collinge 2000; Krauss *et al.* 2003; Ricketts 2001; Summerville & Crist 2001]. Additionally, invertebrates usually have to complete their life cycle every year, which means that they react more rapidly to environmental or management changes than long-living organisms such as most birds, mammals and certain perennial plant species [Mc Geoch 1998; New 1995c]. All the above mentioned characteristics [narrow niche, limited patch size, low mobility, early warning] are major assets of invertebrates in applied conservation practices and make them complementary to the more commonly used vertebrates or vascular plants.

In the light of both vertebrate and invertebrate conservation biology, it is very important to emphasize the difference between biotope and habitat [Dennis *et al.* 2003]. A biotope is a rather general classification of communities [often vegetation types]; habitat refers to species-specific resources and conditions that are necessary for its survival [Hall *et al.* 1997]. Founding the conservation of species, and especially invertebrates, solely on biotope conservation is likely to fail since it strongly overestimates the habitat area that is actually used by the species [Dennis *et al.* 2003; Van Dyck & Vanreusel 2002]. An even better approach would be the so-called functional resource-based habitat concept where the habitat is described as a set of resources and conditions required by a species [Dennis *et al.* 2003]. This resource-based habitat concept not only allows for a more precise definition of species' habitats, it also permits a more optimal use of species information in nature conservation. Communication with people in the field [managers, wardens] is easier when guidelines about species-specific resources [in the case of butterflies e.g., density of host plants, nectar sources, roosting sites, mate location sites] or conditions [e.g., microclimate] can be given than when vague recommendations on the management of the biotope are put forward [Dennis *et al.* 2003]. Even in scientific literature the terms 'biotope' and 'habitat' are often used as synonymes.

Invertebrate conservation biology has mainly focused on the most conspicuous taxa such as butterflies or dragonflies. It is assumed that principles emerging from the study of such relatively well-known taxa are, under certain circumstances, generally applicable to other, less-well known invertebrate taxa [Mc Geoch 1998; New 1995c; New 1997]. Detecting when conservation of a certain invertebrate species or taxon is necessary is not always easy: population sizes of most invertebrates can fluctuate considerably among years and large differences in numbers among years do not necessarily indicate a declining population trend. Long-term monitoring or reliable historical data can help to assess the threat status of invertebrates, but such data are not readily available for invertebrate taxa [except for butterflies in some countries; Pollard & Yates 1993; van Swaay *et al.* 2002].

Why invertebrates?

Invertebrates constitute 75-80% of the global biodiversity, which is estimated at about 13.5 million species [Heywood 1995; Wheeler 1990] and occur in nearly all terrestrial and aquatic environments throughout the world; about 75% of all invertebrates are insects [Samways 1994; Fig. 1.1]. They are important as essential constituents of the communities in all ecosystems outside the polar region [New *et al.* 1995]. The majority of invertebrates are rather small, inconspicuous and unpopular with the general public. This has caused both a taxonomical and an ecological impediment for [the use of] invertebrates in nature conservation [New 1995c; Samways 1993]. The taxonomic impediment relates to the fact that a large part of all existing invertebrates has not been described yet; additionally, species in some invertebrate groups can only be classified by a few experts. Moreover, the geographic distribution of these experts is not proportional to invertebrate species richness: most experts are found in temperate-zone regions where invertebrate diversity is relatively low compared to tropic regions [New 1995c]. The ecological impediment can be referred to as the lack of ecological knowledge for most of the invertebrates which seriously hampers the use of invertebrates in nature conservation [Samways 1994]. Apart for some conspicuous groups such as butterflies, dragonflies and some beetle families, very little is known on the ecology of the majority of invertebrates. Even for butterflies, the most intensively studied invertebrates throughout the world, ecological knowledge is, like taxonomical expertise, unevenly distributed: most detailed autecological studies were/are carried out in Europe and north America while such studies are far less numerous in the much more species-rich tropical regions [Larsen 1995; New 1995a].

Figure 1.1. 'Species-scape' - a landscape of different species that symbolize life on Earth [Wheeler 1990]. Each of the organisms pictured represents a different group of living things, and the size of each picture indicates the known number of species that are in that group, compared with the known numbers in other groups: 1. Bacteria [4,000]; 2. Fungi [72,000]; 3. Algae [40,000]; 4. Trees, shrubs, and other vascular plants [270,000]; 5. Protozoa [40,000]; 6. Sponges [10,000]; 7. Corals, jellyfish, and relatives [10,000]; 8. Flatworms [20,000]; 9. Roundworms [25,000]; 10. Earthworms and relatives [12,000]; 11. Clams, squids, and other molluscs [70,000]; 12. Starfish and relatives [6,100]; 13. Insects [950,000]; 14. Spiders, crustaceans, and other non-insect arthropods [123,400]; 15. Fishes, tunicates, and lancelets [19,000]; 16. Amphibians [4,200]; 17. Reptiles [6,300]; 18. Birds [9,000]; 19. Mammals [4,000].

Invertebrate conservation in Flanders

About 40,000 species are known in Flanders, of which 30,000 [75%] are invertebrates [De Bruyn *et al.* 2003; Peeters *et al.* 2003]. Despite their large numbers and their functional importance in every ecosystem [e.g., ants as soil developers [Hölldobler & Wilson 1990]; bumble bees and honey bees as important pollinators [Carvell 2002], prey, predation ...], legal species protection, ecological research on and/or mapping projects for invertebrates are disproportional to the number of species they represent. The Flemish nature conservation authorities state that, generally speaking, species have already been extensively studied in Flanders [Ministerie van de Vlaamse Gemeenschap, departement Leefmilieu en Infrastructuur [LIN], Administratie Milieu, Natuur, Land- en Waterbeheer [AMINAL] 1997, p. 131]. Distribution atlases do exist or are in progress for a variety of taxonomic groups [Table 1]. For example, vascular plants and all vertebrate groups already have or will have relatively up-to-date atlases in the near future. Invertebrate groups are relatively ill-represented in this list of existing atlases and some of them are certainly outdated. This discrepancy is also present in legal species protection in Flanders, based on the Royal decree of 1980, where 63% of all vertebrate species is strictly protected against only 0.5% of all invertebrate species [De Pue *et al.* 2003]. The same holds true for the Red Lists that are available for all vertebrates and vascular plants in Flanders, while they have been compiled for only 5-6% of the invertebrate species [De Bruyn *et al.* 2003]. The fact that more information is published on vertebrates in Flanders, has lead to an even larger discrepancy in the existing species action plans in Flanders: out of 12 action plans, 11 are for vertebrate species and only one for an invertebrate [the Alcon Blue butterfly *Maculinea alcon*; Table 2].

Table 1.1. Published distribution atlases and Red Lists in Flanders. Distribution atlases or Red Lists marked with an asterix are published as provisional.

Taxonomic group	Distribution atlas	Red List
Fungi [partim]	-	Walleyn & Verbeken [1999]
Vascular plants	van Rompaey & Delvosalle [1979]	Biesbrouck *et al.* [2001]
Landsnails	-	Backeljau *et al.* [in prep.]
Myriapoda	Lock [2000]*	-
Spiders	Alderweireldt & Maelfait [1990] Baert [1996] De Blauwe & Baert [1981] Jacobs [1993] Janssen [1993] Janssen & Baert [1987] Ransy & Baert [1987a,b] Ransy & Baert [1991a,b] Ransy *et al.* [1991] Segers & Baert [1991] Van Keer & Vanuytven [1993]	Maelfait *et al.* [1998]
Butterflies	Maes & Van Dyck [1999]	Maes & Van Dyck [1999]
Carabid beetles	Desender [1986a,b,c,d]	Desender *et al.* [1995]
Dragonflies	-	De Knijf & Anselin [1996]
Ants	Dekoninck *et al.* [2003]*	Dekoninck *et al.* [2003]*
Grasshoppers and crickets	Decleer *et al.* [2000]*	Decleer *et al.* [2000]*
Dolichopodid flies	Pollet [2000]	Pollet [2000]
Hoverflies	Verlinden [1991]	-
Empidid flies	-	Grootaert *et al.* [2001]
Waterbugs	Bonte *et al.* [2001]	Bonte *et al.* [2001]
Fish	Vandelannoote *et al.* [1998]	Vandelannoote *et al.* [1998]
Amphibians & Reptiles	Bauwens & Claus [1996]	Bauwens & Claus [1996]
Breeding birds	Vermeersch *et al.* [in prep.]	Devos & Anselin [1999]*
Mammals	Verkem *et al.* [2004]	Criel [1994]

Table 1.2. Species action plans in Flanders.

Species	Taxonomic group	Dutch name	Reference
Maculinea alcon	Butterflies	Gentiaanblauwtje	Vanreusel *et al.*[2000]
Cottus gobio	Fish	Rivierdonderpad	Seeuws *et al.* [1999a]
Cobitis taenia	Fish	Kleine modderkruiper	Seeuws *et al.* [1999b]
Lampetra planeri	Fish	Beekprik	Seeuws *et al.* [1996]
Hyla arborea	Amphibians	Boomkikker	Vervoort & Goddeeris [1996]
Alytes obstetricans	Amphibians	Vroedmeesterpad	Vervoort [1994]
Vipera berus	Reptiles	Adder	Bauwens *et al.* [1995]
Perdix perdix	Birds	Patrijs	Van Daele & Matthysen [1996]
Cricetus cricetus	Mammals	Hamster	Valk *et al.* [2001]
Meles meles	Mammals	Das	Econnection [1991,1996]
Lutra lutra	Mammals	Otter	Criel [1996]
Chiroptera	Mammals	Vleermuizen	Verkem & Verhagen [2000]

This bias towards vertebrates and vascular plants is certainly not restricted to Flanders. 89% of the species on the annexes I, II or IV in the EU Bird Directive and/or EU Habitat Directive are vertebrates and/or vascular plants while only 11% consists of invertebrates. This descrepancy is even greater on a global scale where only 0.2% of all invertebrates is assigned to a threat category [Critically endangered, Endangered or Vulnerable] and 7% in vertebrates, despite the fact that invertebrates appear to be more threatened than vertebrates [29% of the assessed invertebrates vs. 18% of the assessed vertebrates [http://www.redlist.org/info/tables/table1.html].

Aims

The role of conservation biology is to provide a scientific basis for action and a vital role of the conservation biologist is to effectively communicate the results of scientific research to enable appropriate action to be taken by all people concerned [e.g. wardens, volunteers, policy makers; Pullin 2002a]. This vital step for effective conservation of species on the ground is often forgotten or neglected by scientists [Pullin 2002b]. The effective conservation of threatened species, certainly in regions with an intensive land use such as Flanders, can only be accomplished through a division of tasks between conservation biologists and practitioners. Therefore, it is equally important that managers make an effort to be informed about the real world of science as it is that scientists make an effort to be informed about the real world of management. This means that scientists should be aware of the questions managers have in the field [when to mow?, how many grazers to use?, why is the vegetation changing towards an undesired species composition?, etc.] and of the way policy makers use scientific information to underpin political or social decisions [how to choose between two sites proposed for conservation? what species to protect legally?, etc.]. Scientists should, therefore, be willing to incorporate practical field questions into their research programmes. On the other hand, managers and policy makers should be stimulated to contact conservation biologists with their questions.

The main aim of the present thesis is to explore to what extent the use of [multi-]species information provides an added value to nature conservation and policy making in Flanders. The applied methodologies and the three case studies mainly use butterflies as model organisms. Insects and other invertebrates are increasingly promoted as suitable model organisms to deal with conservation questions because they act on small to intermediate scales and have a wide variety of life history traits [New 1997; Watt & Boggs 2003]. Butterflies have the additional benefit of being popular organisms which facilitates communication with practitioners and even with the general public.

More detailed aims of this thesis are:

1 providing scientific methodologies for analysing existing species information in support of nature conservancy policy making starting from distribution databases: the compilation of Red Lists [**Chapter 2**] and detecting changes in butterfly numbers linked to land use changes [**Chapter 3**];

2 overcome biases in distribution databases by applying modelling techniques to optimise mapping schemes and to facilitate cross-taxon species richness patterns comparisons [**Chapters 4-5**];

3 exploring the added value of explicitly incorporating species information [ants, the Alcon Blue butterfly] in the conservation and management of a threatened biotope [wet heathland] [**Chapters 6-7**];

4 gathering and implementing the necessary information for the protection of a target species, i.e., the endangered Alcon Blue butterfly *Maculinea alcon* in such a threatened biotope [**Chapter 7**];

5 exploring the use of species as tools by applying a multispecies approach in nature conservation applications [**Chapter 8**];

6 translating and applying the results of this scientific research into practical conservation guidelines that are executable by conservation practitioners and/or by policy makers.

Outline

This thesis focuses on the use of [invertebrate] species information in nature conservation in Flanders and on the translation and implementation of the results of scientific research to people in the field [wardens and managers on the one hand and policy makers on the other]. It consists of two parts: the first one deals with different methods of using distribution and ecological data to increase the use of species in nature conservancy policy making. The second part illustrates three case studies of how the incorporation of species-specific knowledge can be used to manage or assess the quality and/or quantity of wet *Erica tetralix* heathland [a biotope of European conservation concern] and how detailed species information can improve the survival of a European-wide threatened species. Fig. 1.2 illustrates how the different chapters of this thesis investigate the possible use of species information in different nature conservation purposes.

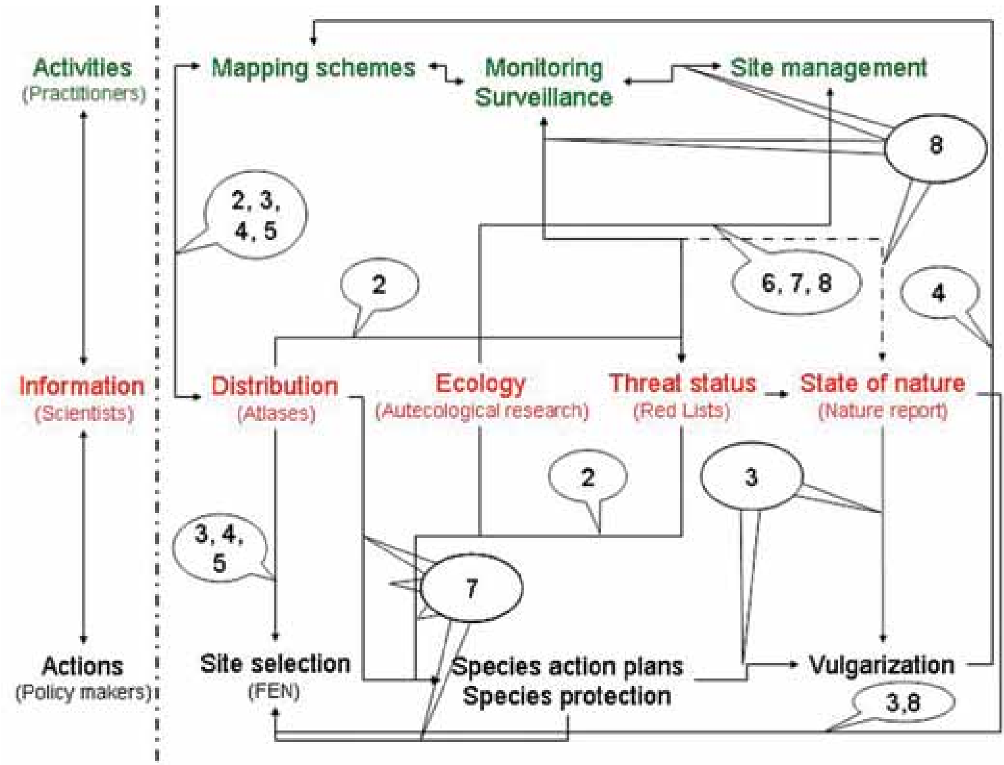

Figure 1.2. Activities, available information and possible actions in nature conservation. The numbers in the text balloons refer to the different chapter numbers in this thesis that are dealing with the use of different types of information in nature conservation.

Species distribution data bases are usually used to publish atlases for a given region. But, such data bases are more than a mere collection of distribution data [Lobo *et al*. 1997; Speight & Castella 2001]. Analyzing them thoroughly can not only increase their utility in nature conservancy policy considerably, but can also make mapping schemes much more efficient [Dennis & Hardy 1999]. One of the possible applications of distribution data with historical and recent observations, is the compilation of Red Lists [Maes *et al*. 1995]. Uniform categories and criteria for Red Lists are of major importance to allow for comparison of threat statuses among species from different taxonomic groups. **Chapter 2** deals with Red List methodology and describes the criteria and categories that are now commonly applied in Flanders and in the Netherlands [Maes & van Swaay 1997]. **Chapter 3** describes a possible methodology for dealing with biases [temporal and geographical] in distribution data bases; here, we analyse the loss of butterfly diversity in Flanders together with the causes of the decline [Maes & Van Dyck 2001]. Butterflies are used here as model organisms to evaluate some aspects of the state of nature in Flanders [cf. Dumortier *et al*. 2003; Kuijken 1999; Kuijken *et al*. 2001]. **Chapter 4** uses modelling techniques to predict butterfly species richness in Belgium [Maes *et al*. 2003]. Such modelling techniques are useful to optimise mapping schemes for two reasons: [1] they can determine the minimum number of mapping grid squares to be surveyed for future atlases and [2] they can incorporate un-surveyed or under-surveyed squares in nature conservancy policy making [Dennis & Hardy 1999; Dennis & Shreeve 2003; Dennis *et al*. 1999, 2002; Dennis & Thomas 2000]. **Chapter 5** [Maes *et al*. in press] extends the previous work and delimits species-rich areas for different taxonomic groups [vascular plants, dragonflies, butterflies, herpetofauna and birds]. By including different taxonomic groups [representing different scales and different trophic levels] in delimiting species-rich sites, it is more justified to assume that other, un-surveyed and/or less conspicuous species groups will be conserved as well.

The second part deals with detailed ecological research on threatened invertebrate species of a habitat of European conservation concern; wet *Erica tetralix* heathland is used as model biotope type to indicate the value of using detailed ecological research on invertebrates for its conservation.

Chapter 6 [Maes *et al*. 2003] describes ant communities on wet heathlands and the negative impact of the encroachment of grasses on the microclimatic conditions. Some specialised and threatened species of wet heathlands are myrmecophilous and are therefore completely depending on the presence and abundance of particular ants. One of the threatened myrmecophilous species is the Alcon Blue butterfly *Maculinea alcon*, a species of European conservation concern [van Swaay & Warren 1999]. **Chapter 7** [Maes *et al*. in press] deals with delimiting 'functional conservation units' for *Maculinea alcon* and describes detailed management measures and priorities for its conservation. The delimitation of functional conservation units within which management measures differ in intensity, facilitates the communication towards people in the field. The conservation efforts put into a single species in nature conservation is often necessary because of its precarious local situation and/or because of its European threat status. The Alcon Blue butterfly *Maculinea alcon* is one of the very few invertebrate species in Flanders that is considered threatened in Europe and therefore also merits conservation efforts that are based on thorough scientific research. Single species conservation, however, does not necessarily imply the conservation of other sympatric species [see also **Chapter 5**]. Therefore, multi-species approaches are nowadays advocated but should be thoroughly tested before application. Too many conservation biologists promote their species or taxonomic group of interest as good bio-indicators [Andersen 1999] but rarely provide evidence for such statements. In **Chapter 8** [Maes & Van Dyck submitted], a so called multi-species approach was developed for wet heathlands in which the information content of easily recognisable species is used as a signalling function for both management and habitat quality evaluations.

Finally, an integrated general discussion [**Chapter 9**] summarizes the use of species information in nature conservancy policies on the one hand and in nature management on the other. Suggestions will be made to enhance the use of existing information on species in nature conservation in Flanders. Furthermore, knowledge gaps that hamper the use of species information in nature conservation will be indicated and priorities for future research will be suggested.

2 Uniformity in Red List compilation

foto: Jeroen Mentens

"Conservation action without good science to underpin is like alchemy, or faith healing. Both sometimes produce desirable results, but you have no idea why, and mostly they don't. And yet there is a fundamental, and frequently unrecognised dilemma at the heart of conservation efforts, which the heat of the battle tends to obscure, rather than to illuminate. What do we wish to conserve, and why?"

John Lawton [1997]. The science and non-science of conservation biology. *Oikos* 97: 3-5.

The compilation of the Red Lists of butterflies in Flanders and The Netherlands was based on two criteria: a trend criterion [degree of decline] and a rarity criterion [actual distribution area]. However, due to the large difference in mapping intensity in the two compared periods, a straightforward comparison of the number of grid cells in which each species was recorded, appeared inappropriate. To correct for mapping intensity we used reference species that are homogeneously distributed over the country, that have always been fairly common and that did not fluctuate, increase or decrease too much during this century. For all resident species a relative presence in two compared periods was calculated, using the average number of grid cells in which these reference species were recorded as a correction factor. The use of a standardised method and of well-defined quantitative criteria makes national Red Lists more objective and easier to re-evaluate in the future and facilitates the comparison of Red Lists among countries and among different organisms. The technique applied to correct for mapping intensity could be useful to other organisms when there is a large difference in mapping intensity between two periods.

Reprinted from Maes D. & van Swaay C.A.M. [1997]. A new methodology for compiling national Red Lists applied on butterflies [Lepidoptera, Rhopalocera] in Flanders [N.-Belgium] and in The Netherlands. *Journal of Insect Conservation* 1: 113-124.

Introduction

Since their conception in 1963 by Sir Peter Scott, Red Lists have been increasingly used as nature conservation tools [Collar 1996]. Red Lists or Red Data Books may have several uses: [1] to set up research programmes for conservation, [2] to derive conservation priorities and [3] to propose protection for sites that are inhabited by threatened species [Collar 1996; Mace 1994]. Their usage stresses that categorisation of the different species should be based on reliable and objective criteria. In the past, almost all Red Lists were compiled on the basis of a best professional judgement by a group of experts. With their introduction for use in the compilation of international Red Lists by the International Union for the Conservation of Nature and Natural Resources [IUCN] [IUCN Species Survival Commission 1994; Mace & Stuart 1994], quantitative criteria are slowly finding their way into national Red Lists as well [e.g., Schnittler *et al.* 1994 in Germany]. However, since much more data are available on vertebrates and on vascular plants, the proposed IUCN criteria are more easily applicable to these groups than to lower organisms, such as invertebrates or lower plants [Hallingbäck *et al.* 1995].

The method proposed by Stroot & Depiereux [1989] for compiling the Red List of the Trichoptera in Belgium and which is based on the χ^2-distribution, cannot be applied to the data set of the butterflies in Flanders and The Netherlands. In order to use their method, the chance of finding a species should be equal in both compared periods; this condition is certainly not fulfilled since in the past more emphasis was on recording rare species while nowadays the common species represent the majority of the records. Recently, Avery *et al.* [1995] proposed another method for compiling the national Red List of British birds. The combined use of three axes [axis 1 = the national threat status, axis 2 = the international importance and axis 3 = the European/global conservation status] were used as the basis for setting UK conservation priorities. However, due to lack of sufficient data, their method is difficult to use for invertebrates and in that case, they propose the use of qualitative information. Since the IUCN proposed a new approach for compiling Red Lists, it is recommended to work out methods that use quantitative criteria, even for invertebrates or other lower organisms.

In Flanders [N.-Belgium] and The Netherlands, Maes *et al.* [1995] and van Ommering [1994] recently proposed categories and criteria for the compilation of the respective national Red Lists. Although it is only a region of Belgium, we apply the terms “country” and “national” for Flanders to simplify the writing. The principal idea in this new method to compile national Red Lists is that the present rarity of a species is compared with its rarity in a reference period. The distribution area in the reference period is considered as being the more or less natural distribution of most species. In The Netherlands, a lot of butterflies showed a marked and strong decrease in the period 1950-1980 [van Swaay 1990]. In this period the Dutch landscape lost many suitable butterfly habitats due to the intensification of agriculture, acidification, etc. Therefore, the year 1950 marks the end of the reference period in The Netherlands. The start in 1901 was chosen arbitrarily. The number of butterfly records before this year was very low.

The method proposed for the compilation of the Red Lists in Flanders and The Netherlands uses a combination of the actual rarity and the degree of decline in distribution area to assign all resident species to a Red List category. The actual rarity is expressed as the extent of the present day distribution area and is measured as the number of grid cells wherein a species was recorded in the period 1981-1995 in Flanders and the period 1986-1993 in The Netherlands [= period 2]. This is a fairly straightforward procedure. The second criterion compares the present day distribution area with that in the period 1901-1980 in Flanders and 1901-1950 in The Netherlands [= period 1]. Due to the large difference in mapping intensity between the two compared periods, we had to work out a way to compensate for this difference. In this paper we describe the general methodology for compiling the

Red Lists in Flanders and The Netherlands. In particular, we introduce a technique that corrects for differences in mapping intensity among sampling periods. This technique may also be used to compare distribution areas of other groups of organisms when there is a large difference in mapping intensity between two sampling periods. The use of a standardised method with well-defined quantitative criteria, such as the one we propose in this paper, makes national Red Lists more objective and easier to re-evaluate in the future and facilitates the comparison of Red Lists among countries as well as among different groups of organisms.

Methods

The data for compiling the Red Lists of Flanders and The Netherlands were gathered by the Flemish Butterfly Study Group and by the Dutch Butterfly Conservation respectively. At first, we gathered data from the literature and from museum and private collections. These data mainly date from before 1980 and comprise about 16,000 records in Flanders and about 125,000 in The Netherlands. Afterwards, both countries organised intensive campaigns with the help of numerous volunteers which greatly increased the data set. In Flanders, this butterfly mapping scheme started in 1991 and the complete data set now comprises about 145,000 records on 69 resident species. In The Netherlands, the mapping project started in 1981 and the complete data set now contains about 430,000 records on 70 resident species [Wynhoff & van Swaay 1995]. As the basis for mapping the distribution of each species, we used grid cells of 5 km x 5 km both in Flanders [UTM projection, n = 636] and The Netherlands [Amersfoort projection, n = 1677].

Red List categories in Flanders and The Netherlands

The Red List categories in Flanders and The Netherlands are based on those of the IUCN Species Survival Commission 1994] and are given in Table 2.1. Both national Red Lists only refer to resident species, present in the country throughout the year and known to reproduce in the wild over a period of at least ten years. Thus, we excluded migratory species such as *Vanessa atalanta* [Red Admiral], *Cynthia cardui* [Painted Lady], *Colias hyale* [Pale Clouded Yellow] and *Colias crocea* [Clouded Yellow]. We used two criteria to classify species into the Red Lists of Flanders and The Netherlands: a rarity criterion and a trend criterion [Table 2.2].

The rarity criterion is defined by the number of grid cells in which a species was recorded in period 2. The limits that determine rarity are arbitrarily chosen. For rare but fairly mobile species [e.g., *Aporia crataegi* [Black-veined White], *Argynnis paphia* [Silver-washed Fritillary], *Issoria lathonia* [Queen of Spain Fritillary], *Leptidea sinapis* [Wood White], *Nymphalis polychloros* [Large Tortoiseshell] and *N. antiopa* [Camberwell Beauty]], grid cells with single, vagrant individuals were excluded for compiling the Red Lists since they do not concern populations.

Table 2.1. Red List categories and criteria used in The Netherlands and Flanders based on the new IUCN criteria [IUCN Species Survival Commission 1994].

Red List category	Description
Extinct in the wild in Flanders/ The Netherlands [EXF/EXN]	Species that did not have reproducing populations in Flanders/The Netherlands the last ten years. Some of these species are still observed as vagrants.
Critically endangered [CE]	Very rare species that decreased by at least 75% in distribution area between the two compared periods. In Flanders species that have only a few isolated populations also qualify for this category.
Endangered [EN]	Very rare species that have decreased in distribution area by 50-75% between the two compared periods or rare species that have decreased by at least 50% in distribution area between the two compared periods.
Vulnerable [VU]	Very rare or rare species that have decreased in distribution area by 25-50% between the two compared periods or fairly rare species that have decreased in distribution area by at least 25% between the two compared periods.
Susceptible [SU]	Very rare species that have decreased in distribution area by less than 25% between the two compared periods [subcategory "Rare" in Flanders] or common species that have decreased in distribution area by at least 50% between the two compared periods [subcategory "Near-threatened" in Flanders].
Data deficient [DD]	Species of which their are insufficient data to categorise them into a Red List category.
Safe/Low risk [S/LR]	Rare and fairly rare species that have decreased in distribu tion area by less than 25% between the two compared peri ods or common species that have decreased in distribution area by less than 50% between the two compared periods.

The trend criterion is derived from the comparison between the actual rarity of a species with the extent of its distribution area in the past, expressed as the number of grid cells in which a species was recorded in period 1. However, due to the large difference in mapping intensity between past and present, a simple comparison of the number of grid cells in the two periods is inappropriate. In Flanders there are about 13,000 records from the first period and about 130,000 from the second period, while in The Netherlands respectively 42,000 and 260,000 records are available. Furthermore, in the first period mostly rare butterflies were collected or reported in literature, while after 1981 all species were recorded. To tackle

Table 2.2. Classification scheme for the Red Lists of Flanders and The Netherlands. The number of grid cells that determine rarity are arbitrarily chosen.

Presence and percentage of grid cells				
	Very rare < 1%	Rare 1-5%	Fairly rare 5-12.5%	Common >12.5%
Number of grid cells Flanders				
	1-6	7-32	33-80	>80
Number of grid cells The Netherlands				
	1-17	18-83	84-209	>209
Decline in distribution area between the two compared periods [%]				
76-100%	Critically endangered	Endangered	Vulnerable	Susceptible
51-75%	Endangered	Endangered	Vulnerable	Susceptible
26-50%	Vulnerable	Vulnerable	Vulnerable	Safe/Low risk
≤ 25%	Susceptible	Safe/Low risk	Safe/Low risk	Safe/Low risk

the problem of the large difference in mapping intensity in the two compared periods, we use reference species to calculate a relative presence for each species in both periods. The decline in distribution area, calculated with the relative presence's, will then be used as a trend criterion.

Red List categories in Flanders and The Netherlands

For determining reference species, we used a method proposed by van Latour & van Swaay [1992] that was already applied to determine the changes in butterfly abundance's in The Netherlands [van Swaay 1995].

First, for all resident species the number of grid cells in which it was observed was counted per pentad [= period of five years; pentad 1 = 1901-1905, pentad 2 = 1906-1910, etc.]. We subsequently expressed the number of grid cells in which a species was observed per pentad as a percentage of the total number of mapped grid cells in that pentad by [1]

[1]

$$pp_{i,p} = 100 \times \frac{x_{i,p}}{n_p}$$

where $pp_{i,p}$ is the presence in percentage of species i in pentad p, $x_{i,p}$ is the number of grid cells in which species i was recorded in pentad p and n_p is the total number of mapped grid cells [i.e., grid cells whit at least one observation] in pentad p. Secondly, we regressed the presence in percentage against pentad number for those species that are presently common, i.e., that were recorded in at least half of the total number of grid cells, and that are homogeneously distributed over the country. We applied this linear regression only for the periods before which the intensive mapping schemes started: up to and including pentad 18 [1986-1990] in Flanders and up to and including pentad 16 [1976-1980] in The Netherlands. Mapping intensity was considered more or less equal before the beginning of the intensive mapping schemes in both countries.

Reference species should then fulfil the following criteria: [1] the species should not have fluctuated too much during this century [i.e., the coefficient of determination $R^2 \geq 0.20$], [2] the species should have been observed in at least 10% of the mapped grid cells in the beginning of this century [i.e., the intercept on the Y-axis $a \geq 10$] and [3] the species should not have increased or decreased too strongly during this century [i.e, $-1 <$ regression slope $b < +1$]. The habitat in which reference species occur is not taken into account.

Using reference species to compile the Red List

As a measure of the mapping intensity during the periods 1 and 2, the average number of grid cells in which the reference species were recorded in these two periods, was calculated as [2]

[2]

$$\bar{r}_j = \frac{\sum_{t=1}^{n_r} x_{t,j}}{n_r}$$

where $\bar{r}_j$ is the average number of grid cells in which all reference species were recorded in period j, $x_{t,j}$ is the number of grid cells in which reference species t was recorded in period j and n_r is the total number of reference species. Using the average number of grid cells in which the reference species were recorded, we corrected for mapping intensity in both periods by calculating a relative presence for each species by [3]

[3]

$$rp_{i,j} = 100 \times \frac{x_{i,j}}{\bar{r}_j}$$

where $rp_{i,j}$ is the relative presence of species i in period j, $x_{i,j}$ is the number of grid cells in which species i was recorded in period j and $\bar{r}_j$ is the average number of grid cells in which the reference species were recorded in period j. By using the relative presence's in both periods, the decline in distribution area for all resident species was estimated by [4]

[4]

$$d_i = \left[100 \times \frac{rp_{i,2}}{rp_{i,1}}\right] - 100$$

where d_i is the decline in distribution area of species i, $rp_{i,1}$ is the relative presence of species i in period 1 and $rp_{i,2}$ is the relative presence of species i in period 2.

Using the number of grid cells in which a species was recorded in period 2 [$x_{i,2}$] as a rarity criterion and the decline in distribution area [d_i] as a trend criterion, we classified all resident butterfly species into the Red Lists of Flanders and The Netherlands according to the scheme in Table 2.2.

Results

The results of the linear regression analyses applied on the species presence in percentage per pentad are shown in Table 2.3. We determined three reference species in both countries: *Lasiommata megera* [Wall Brown], *Lycaena phlaeas* [Small Copper] and *Polyommatus icarus* [Common Blue] in Flanders and *Coenonympha pamphilus* [Small Heath], *L. phlaeas* [Small Copper] and *Maniola jurtina* [Meadow Brown] in The Netherlands.

With Equation [2], we calculated the average number of grid cells in which the reference species were recorded in the first and second period: in Flanders $\bar{r}1$ is 154 and $\bar{r}2$ is 379 and in The Netherlands $\bar{r}1$ and $\bar{r}2$ are 238 and 750 respectively. With Equations [3] and [4] we subsequently calculated the relative presence's and the declines in distribution area of all resident butterfly species [Appendix 2.1]. According to the scheme in Table 2.2, we then assigned all species to a Red List category [Appendix 2.1].

The use of these criteria results in 20 [29%] and 17 [24%] species in the "Extinct" category and a further 25 [36%] and 30 [43%] species are considered threatened [categories "Critically endangered", "Endangered", "Vulnerable" and "Susceptible"] on the Red Lists in Flanders [Maes & Van Dyck 1996] and The Netherlands [Wynhoff & van Swaay 1995] respectively. In both countries 23 species are considered as not threatened presently [Table 2.4].

Table 2.3. Results of the linear regression on the species presence in percentage. R^2= coefficient of determination, a = intercept on the Y-axis, b = regression slope. When figures are in bold they fulfil the criterion for reference species.

		Flanders			The Netherlands		
Scientific name	Dutch name	R^2	*a*	*b*	R^2	*a*	*b*
Aglais urticae	Kleine vos	**0.56**	-1.1	2.13	**0.78**	-5.3	1.67
Araschnia levana	Landkaartje	**0.67**	-7.6	2.02	**0.51**	-5.1	1.55
Celastrina argiolus	Boomblauwtje	**0.22**	8.9	**0.71**	0.09	**11.8**	0.18
Coenonympha pamphilus	Hooibeestje	**0.61**	4.7	1.22	**0.57**	**11.9**	**0.71**
Gonepteryx rhamni	Citroenvlinder	**0.48**	2.2	1.33	**0.75**	4.3	1.03
Inachis io	Dagpauwoog	**0.60**	-2.4	2.06	**0.71**	-3.5	1.42
Lasiommata megera	Argusvlinder	**0.26**	**9.7**	**0.77**	**0.57**	6.29	**0.78**
Lycaena phlaeas	Kleine vuurvlinder	**0.30**	**12.1**	**0.86**	**0.29**	**14.9**	**0.39**
Maniola jurtina	Bruin zandoogje	**0.34**	8.3	**0.83**	**0.28**	**13.7**	**0.30**
Pararge aegeria	Bont zandoogje	**0.42**	3.7	1.62	-	-	-
Pieris brassicae	Groot koolwitje	**0.48**	1.6	1.43	**0.93**	-2.9	1.27
Pieris napi	Klein geaderd witje	**0.31**	**11.5**	1.26	**0.90**	-1.9	1.29
Pieris rapae	Klein koolwitje	**0.43**	3.5	1.70	**0.89**	-3.7	1.51
Polygonia c-album	Gehakkelde aurelia	**0.56**	-2.5	1.51	-	-	-
Polyommatus icarus	Icarusblauwtje	**0.20**	**14.3**	**0.69**	0.05	**17.7**	**0.15**
Thymelicus lineola	Zwartsprietdikkopje	**0.74**	-1.4	1.08	**0.43**	6.0	**0.35**

Table 2.4. Number of species and percentage per Red List category in Flanders and in The Netherlands.

	Flanders	The Netherlands
Extinct	20 [29%]	17 [24%]
Critically endangered	8 [12%]	7 [10%]
Endangered	6 [9%]	11 [16%]
Vulnerable	7 [10%]	10 [14%]
Susceptible	4 [6%]	2 [3%]
Data deficient	1 [1%]	-
Safe/Low risk	23 [33%]	23 [33%]
Total number of resident species	69	70

Discussion

The classification of the resident butterfly species in Flanders and The Netherlands into the national Red Lists, using the proposed method, lead to useful results for national nature conservation purposes. All butterflies listed as threatened on both Red Lists are indeed specialists of typical habitats that need urgent protection in Flanders and The Netherlands. The same classification method has already been successfully applied for compiling national Red Lists of a wide variety of other organisms like carabid beetles [Desender *et al.* 1995], amphibians and reptiles [Bauwens & Claus 1996] and dragonflies [De Knijf & Anselin 1996] in Flanders and mammals [Hollander & van der Reest 1994], birds [Osieck & Hustings 1994] and grasshoppers [Odé 1997] in The Netherlands.

Criteria like rarity and decline are used in most Red Lists like the British Red

Data Books [Bratton 1991; Shirt 1987], but decline is usually described in a qualitative way ["rapid", "continuous", etc.]. In the newly proposed IUCN criteria [Mace & Stuart 1994], the decline and the rarity criterion are used independently from one another: a species that has either declined in distribution area with at least 80% or that is very rare, is categorised as being "Critically endangered". Adopting the IUCN criteria to the national Red Lists of Flanders and of The Netherlands would have placed respectively 14 and 15 species in the "Critically endangered" category, 7 and 12 species in the "Endangered" category and 1 and 6 species in the "Vulnerable" category. The additional criteria [the degree of potential immigration to counteract the decline] that the IUCN proposed for applying Red List categories at the national level [agreed at the National Red List Workshop in Gland, Switzerland, 23-24 March 1995] are difficult to apply for butterflies. Although some of the threatened or extinct butterflies are potentially fairly mobile, they do not seem to be able to found new populations in our countries. In Flanders and The Netherlands [but also in Germany [Schnittler *et al.* 1994]], the combined usage of the decline and rarity criteria, resulted in a classification into Red List categories on a national level that corresponded better with the authors judgements on butterfly threats in both countries than when IUCN criteria would have been used.

Method for correcting for mapping intensity

Our method first identifies reference species, that will consequently be used to calculate a decline in distribution area. Since reference species should be homogeneously distributed over the country, it is not surprising that only grassland species qualify as reference species as grasslands are the only habitats that are homogeneously distributed in both countries. Furthermore, these species are best represented in the families of the *Lycaenidae* and the *Satyrinae*. The fact that the reference species are only found among grassland species strictly means that this method should only be used to evaluate the change in distribution area of grassland species. For species from other habitats, this method requires the additional assumption that butterflies in other habitats [e.g., forests, heathlands, etc.] were mapped with a similar effort as those in grasslands during both compared periods.

In most European countries, 10 km x 10 km UTM grid cells are used for mapping invertebrates [e.g., Desender 1986a, b, c, d; Emmet & Heath 1989; Geijskes & van Tol 1983]. The large number of data in Flanders and The Netherlands made mapping possible on a 5 km x 5 km scale. The imprecision of the older data [where often only the name of a town or an approximate location is given] did not allow the use of a finer scale. In Flanders, species that declined in distribution area on the basis of 5 km x 5 km grid cells also did so when 10 km x 10 km grid cells were used [$r = 0.951$, $n = 67$, $p < 0.001$]. The use of 5 km x 5 km grid cells, instead of the usual 10 km x 10 km grid cells, certainly allowed a better estimation of the decline in distribution area, but for most species we still underestimated the decline. Since, declines on distribution maps are only detected when all populations have disappeared from a grid cell [Thomas & Abery 1995]. The use of 10 km x 10 km grid cells in Flanders instead of the 5 km x 5 km grid cells, would have underestimated the decline of the rare species for 4% on average and for 36% on average for the intermediately rare species [cf. Thomas & Abery 1995].

The method applied here to correct for mapping intensity, yielded informative results for the butterflies in Flanders and The Netherlands and proved to be useful for other groups of organisms that have been relatively well recorded throughout this century. This technique allowed a fairly good estimation of the decline in distribution area of rare and intermediately rare species, but not for the very common species. This is due to the fact that the latter were largely under-recorded in the past. Since we were compiling a list of threatened species, used to set conservation priorities in Flanders and The Netherlands, the presently common species were of a lesser concern for this purpose. For species with a very localised distribution area within both countries and which were recorded very well in the past, this method calculated a large decline in distribution area by correcting for mapping intensity [e.g., a decline of 73% and 59% for *Cupido minimus* and *Heteropterus morpheus* respectively in Flanders or 75% and 68% for *Boloria aquilonaris* and *Vacciniina optilete* respectively in The Netherlands]. Most of these species inhabit typical and very localised habitats [chalk grasslands, peat bogs, etc.] and data suggest that their distribution area did not undergo changes. Species in such cases are classified in the subcategory "Rare" of the Red List category "Susceptible" in Flanders because of their restricted distribution area in both past and present.

Comparing the Red Lists of Flanders and The Netherlands

The method we used to compile our Red Lists is repeatable and fairly objective. Furthermore, by using the same classification technique in Flanders and The Netherlands, their respective Red Lists become more easily comparable. However, the category "Susceptible" has to be interpreted differently in Flanders and The Netherlands. The four species in this category in Flanders have always had a restricted and localised distribution and are therefore put in the subcategory "Rare". The two species in this category in The Netherlands on the other hand, are still common but have decreased in distribution area by at least 50%. A second difference between both Red Lists is that the reference periods are not identical [1901-1980 vs. 1981-1995 in Flanders and 1901-1950 vs. 1986-1993 in The Netherlands]. However, this does not effect the composition of the Red Lists: by applying the reference periods from The Netherlands to the data of Flanders, we obtained exactly the same Red List for Flanders as with the presently used periods. Since national Red Lists are used for shaping national public policy [Bean 1987], each country can set different but appropriate reference periods.

Comparing the Red Lists of Flanders and The Netherlands, shows that the group of threatened species is almost identical in both countries. Only two species were categorised differently: *Callophrys rubi* is "Vulnerable" in Flanders but "Safe/Low Risk" in The Netherlands while *Papilio machaon* is "Susceptible" in The Netherlands but "Safe/Low Risk" in Flanders. For the species both countries have in common, the degree of decline is very similar [decline in distribution area in Flanders vs. The Netherlands, $r = 0.809$, $n = 63$, $p < 0.001$]. This fact is not surprising since both countries have a similar landscape and have undergone similar declines in the number of suitable butterfly habitats [heathlands, forest, nutrient-poor unimproved grasslands] through changes in agricultural management and building activities. Fragmentation of suitable habitats can strongly decrease or even stop the exchange of individuals between populations leading to a higher risk of extinction [e.g., Thomas & Jones 1993]. Furthermore, a lot of butterfly habitats have deteriorated qualitatively through bad management or lack of management. A management plan for threatened butterflies, both on the population and on the landscape level, has already been produced in The Netherlands [Ministerie van Landbouw, Natuurbeheer en Visserij 1990] and is being prepared for Flanders [Maes and van Dyck in prep.].

A comparison of our Red Lists of butterflies with those in other north-western European countries or regions [not compiled with the new IUCN criteria] reveals that the group of extinct and threatened species in most countries varies from 52% [91 species] in Germany [Pretscher 1984], over 63% [80 species] in Baden-Württemberg [Ebert & Rennwald 1993b] to 66% [51 species] in Wallonia, South-Belgium [Goffart *et al.* 1992]. In Great Britain only 18% [10 species] of the species is extinct or threatened [Shirt 1987]. Although the global figures are alike, except for Great Britain, the proportion of extinct species is clearly higher in Flanders [29%] and in The Netherlands [24%] than in the other countries or regions. With sixteen extinct species [16%], Wallonia [South-Belgium] is intermediate between our countries and the other European countries or regions; Germany with only two [1%], Baden-Württemberg with only four [3%] and Great Britain with only three extinct species [5%] do much better on this point. A comparison of threatened butterflies among countries is difficult due to different techniques used for compiling the national Red Lists. It would be interesting to apply our technique to existing data sets in other countries or regions. Only by using the same technique will national Red Lists become comparable. Since a European Red List is being prepared an objective and repeatable method, like the one proposed here, could be recommended.

Future Red Lists

Since butterfly distribution and threats are variable, Red Lists will have to be updated regularly [e.g., every 10 years]. Thanks to the large number of records that are yearly gathered by numerous volunteers, the distribution of butterflies in Flanders and The Netherlands can be easily monitored now. The next Red Lists in both countries could, for example, compare the distribution of the species in the period 1991-2000 with that in the period 2001-2010. Due to the similar collecting technique [direct observations] and probably fairly similar mapping intensities, the number of grid cells of each species in both periods will be more easily comparable. Harmonisation of the change-over date in future Red Lists should be aimed for throughout Europe and the year 2000 could be ideal for this purpose.

In the future the Butterfly Monitoring Scheme in Flanders and The Netherlands, based on transect counts [Pollard & Yates 1993], might be used in addition to the method proposed in this article, in order to take the trends in the numbers of individuals in the monitored populations of threatened butterfly species into account [van Swaay *et al.* 1997].

Appendix 2.1. Number of grid cells in which the species was recorded in the periods 1901-1980 in Flanders and 1901-1950 in The Netherlands [x_1] and 1981-1995 in Flanders and 1986-1993 in The Netherlands [x_2] and their relative presence in both periods [rp_1, 100% = 154 in Flanders and 238 in The Netherlands; rp_2, 100% = 379 in Flanders and 750 in The Netherlands], the decline in distribution area [d, in percentage] and the Red List category [RLC]. - = the species is not indigenous; v = all observations concern vagrant individuals; [x] = the number of grid cells with reproducing populations is given between brackets, the major part of the observations concerns vagrant individuals; i = re-introduced species. For the abbreviations of the Red List categories we refer to Table 2.1.

	Flanders						The Netherlands					
Species	x_1	x_2	rp_1	rp_2	d	RLC	x_1	x_2	rp_1	rp_2	d	RLC
Aglais urticae	149	542	96.8	143.0	48	S/LR	101	1008	42.4	134.4	217	S/LR
Anthocharis cardamines	111	381	72.1	100.5	40	S/LR	161	518	67.7	69.1	2	S/LR
Apatura ilia	0	1	0	0.3	-	CE	-	-	-	-	-	-
Apatura iris	14	12	9.1	3.2	-65	EN	31	28	13.0	3.7	-71	EN
Aphantopus hyperantus	92	239	59.7	63.1	6	S/LR	149	428	62.6	57.1	-9	S/LR
Aporia crataegi	30	19[V]	19.5	5.0	-74	EXF	98	16[V]	41.2	2.1	-95	EXN
Araschnia levana	101	434	65.6	114.5	75	S/LR	73	694	30.7	92.5	202	S/LR
Argynnis paphia	30	21[1]	19.5	5.5	-72	CE	59	28[V]	24.8	3.7	-85	EXN
Aricia agestis	35	59	22.7	15.6	-32	VU	107	149	45.0	19.9	-56	VU
Boloria aquilonaris	-	-	-	-	-	-	9	7	3.8	0.9	-75	CE
Brenthis ino	-	-	-	-	-	-	5	0	2.1	0	-100	EXN
Callophrys rubi	53	56	34.4	14.8	-57	VU	115	212	48.3	28.3	-42	S/LR
Carcharodus alceae	14	0	9.1	0	-100	EXF	-	-	-	-	-	-
Carterocephalus palaemon	38	64	24.7	16.9	-32	VU	44	65	18.5	8.7	-53	EN
Celastrina argiolus	115	366	74.7	96.6	29	S/LR	166	707	69.8	94.3	35	S/LR
Clossiana euphrosyne	13	0	8.4	0	-100	EXF	31	0	13.0	0	-100	EXN
Clossiana selene	51	1	33.1	0.3	-99	CE	175	53	73.5	7.1	-90	EN
Coenonympha arcania	3	0	2.0	0	-100	EXF	14	2	5.9	0.3	-95	CE
Coenonympha hero	4	0	2.6	0	-100	EXF	4	0	1.7	0	-100	EXN

	Flanders						The Netherlands					
Species	x_1	x_2	rp_1	rp_2	d	**RLC**	x_1	x_2	rp_1	rp_2	d	**RLC**
Coenonympha pamphilus	156	328	101.3	86.5	-15	S/LR	245	742	102.9	98.9	-4	S/LR
Coenonympha tullia	16	5	10.4	1.3	-87	CE	73	18	30.7	2.4	-92	EN
Cupido minimus	6	4	3.9	1.1	-73	SU	8	0	3.4	0	-100	EXN
Cyaniris semiargus	64	2[1]	41.6	0.5	-99	CE	57	1[V]	24..0	0.1	-99	EXN
Erynnis tages	29	2[V]	18.8	0.5	-97	EXF	64	2	26.9	0.3	-99	CE
Eurodryas aurinia	20	0	13.0	0	-100	EXF	64	0	26.9	0	-100	EXN
Fabriciana adippe	9	0	5.8	0	-100	EXF	-	-	-	-	-	-
Fabriciana niobe	7	0	4.6	0	-100	EXF	76	41	31.9	5.5	-83	EN
Gonepteryx rhamni	129	444	83.8	117.2	40	S/LR	174	892	73.1	118.9	63	S/LR
Heodes tityrus	91	4[V]	59.1	1.1	-98	EXF	191	146	80.3	19.5	-76	VU
Hesperia comma	29	22	18.8	5.8	-69	EN	101	98	42.4	13.1	-69	VU
Heteropterus morpheus	5	5	3.3	1.3	-59	SU	6	14	2.5	1.9	-26	VU
Hipparchia semele	82	79	53.3	20.8	-61	VU	179	270	75.2	36.0	-52	SU
Hipparchia statilinus	5	0	3.3	0	-100	EXF	10	16	4.2	2.1	-49	VU
Inachis io	144	543	93.5	143.3	53	S/LR	87	1003	36.6	133.7	266	S/LR
Issoria lathonia	69	25[2]	44.8	6.6	-85	EXF	199	90	83.6	12.0	-86	VU
Ladoga camilla	50	55	32.5	14.5	-55	VU	104	95	43.7	12.7	-71	VU
Lasiommata megera	146	347	94.8	91.6	-3	S/LR	188	825	79.0	110.0	39	S/LR
Leptidea sinapis	12	8[1]	7.8	2.1	-73	CE	-	-	-	-	-	-
Limenitis populi	8	0	5.2	0	-100	EXF	9	3	3.8	0.4	-89	CE
Lycaeides idas	4	0	2.6	0	-100	EXF	14	0	5.9	0	-100	EXN
Lycaena dispar	-	-	-	-	-	-	15	6	6.3	0.8	-87	CE
Lycaena hippothoe	0	1	0	0.3	-	CE	22	0	9.2	0	-100	EXN
Lycaena phlaeas	150	388	97.4	102.4	5	S/LR	237	742	99.6	98.9	-1	S/LR
Maculinea alcon	25	23	16.2	6.1	-63	EN	58	89	24.4	11.9	-51	VU

	Flanders						The Netherlands					
Species	x_1	x_2	rp_1	rp_2	d	**RLC**	x_1	x_2	rp_1	rp_2	d	**RLC**
Maculinea alcon arenaria	-	-	-	-	-	-	5	0	2.1	0	-100	EXW
Maculinea arion	-	-	-	-	-	-	9	0	3.8	0	-100	EXN
Maculinea nausithous	-	-	-	-	-	-	14	2[i]	5.9	0.3	-95	EX[N1]
Maculinea teleius	9	0	5.8	0	-100	EXF	17	2[i]	7.1	0.3	-96	EX[N1]
Maniola jurtina	133	414	86.4	109.2	27	S/LR	233	765	97.9	102.0	4	S/LR
Melanargia galathea	7	18[1]	4.6	4.8	5	SU	-	-	-	-	-	-
Melitaea cinxia	37	6[4]	24.0	1.6	-93	CE	63	1	26.5	0.1	-99	CE
Melitaea diamina	6	0	3.9	0	-100	EXF	18	0	7.6	0	-100	EXN
Mellicta athalia	21	0	13.6	0	-100	EXF	84	20	35.3	2.7	-92	EN
Mesoacidalia aglaja	25	6v	16.2	1.6	-90	EXF	97	27	40.8	3.6	-91	EN
Nordmannia ilicis	53	40	34.4	10.6	-69	VU	115	96	48.3	12.8	-74	VU
Nymphalis antiopa	34	18v	22.1	4.8	-79	EXF	94	15v	39.5	2.0	-95	EXN
Nymphalis polychloros	65	40[10?]	42.2	10.6	-75	EN	139	30	58.4	4.0	-93	EN
Ochlodes venatus	122	312	79.2	82.3	4	S/LR	174	503	73.1	67.1	-8	S/LR
Papilio machaon	126	310	81.8	81.8	0	S/LR	204	248	85.7	33.1	-61	SU
Pararge aegeria	134	493	87.0	130.1	50	S/LR	135	513	56.7	68.4	21	S/LR
Pieris brassicae	138	493	89.6	130.1	45	S/LR	88	873	37.0	116.4	215	S/LR
Pieris napi	165	525	107.1	138.5	29	S/LR	102	965	42.9	128.7	200	S/LR
Pieris rapae	153	558	99.4	147.2	48	S/LR	81	1011	34.0	134.8	296	S/LR
Plebejus argus	63	40	40.9	10.6	-74	VU	111	191	46.6	25.5	-45	VU
Polygonia c-album	110	439	71.4	115.8	62	S/LR	141	576	59.2	76.8	30	S/LR
Polyommatus icarus	167	402	108.4	106.1	-2	S/LR	267	651	112.2	86.8	-23	S/LR
Pyrgus armoricanus	3	0	2.0	0	-100	EXF	-	-	-	-	-	-
Pyrgus malvae	42	11	27.3	2.9	-89	EN	132	38	55.5	5.1	-91	EN
Pyronia tithonus	99	358	64.3	94.5	47	S/LR	146	451	61.3	60.1	-2	S/LR

	Flanders						The Netherlands					
Species	x_1	x_2	rp_1	rp_2	d	**RLC**	x_1	x_2	rp_1	rp_2	d	**RLC**
Quercusia quercus	52	102	33.8	26.9	-20	S/LR	108	306	45.4	40.8	-10	S/LR
Satyrium w-album	17	1	11.0	0.3	-98	DD	11	1	4.6	0.1	-97	CE
Spialia sertorius	3	1	2.0	0.3	-87	SU	7	1v	2.9	0.1	-95	EXN
Thecla betulae	25	22	16.2	5.8	-64	EN	54	28	22.7	3.7	-84	EN
Thymelicus acteon	-	-	-	-	-	-	15	0	6.3	0	-100	EXN
Thymelicus lineola	87	359	56.5	94.7	68	S/LR	136	628	57.1	83.7	47	S/LR
Thymelicus sylvestris	52	165	33.8	43.5	29	S/LR	137	288	57.6	38.4	-33	S/LR
Vacciniina optilete	-	-	-	-	-	-	4	4	1.7	0.5	-68	EN

3 Analyzing biased distribution data bases for conservation purposes

foto: Jeroen Mentens

"The unique symbiosis between informed amateurs/collectors and dedicated professionals in gathering data, performing management, and monitoring the results of conservation programs is one of the great strengths of butterfly conservation. This partnership must continue and incorporate the growing number of butterfly watchers for the benefit of these most charismatic of insects."

Tim New *et al.* [1995]. Butterfly conservation management. *Annual Review of Entomology* 40: 57-83.

We illustrate the strong decrease in the number of butterfly species in Flanders [north Belgium] in the 20th century using data from a national butterfly mapping scheme. Nineteen of the 64 indigenous species went extinct and half of the remaining species are threatened at present. Flanders is shown to be the region with the highest number of extinct butterflies in Europe. More intensive agriculture practices and expansion of house and road building increased the extinction rate more than eightfold in the second half of the 20th century. The number of hotspots decreased considerably and the present-day hotspots are almost exclusively in the Northeast of Flanders. Species with low dispersal capacities and species from oligotrophic habitats decreased significantly more than mobile species or species from eutrophic habitats. We discuss these results in a NW-European context and focus on concrete measures to preserve threatened butterfly populations in Flanders.

Reprinted from Maes D. & Van Dyck H. [2001]. Butterfly diversity loss in Flanders [north Belgium]: Europe's worst case scenario? *Biological Conservation* 99: 263-276.

Introduction

In contrast with many other invertebrates, butterflies have been studied and extensively collected by amateur entomologists and scientists in the past. Particularly in Europe, catalogues on the occurrence of different species have been published since the beginning of the 19th century [e.g., De Selys-Longchamps [1837] for Belgium]. Ever since, interest in butterflies has only increased and they are among the best-known groups and one of the most frequent conservation targets amongst invertebrates [New 1997]. Many butterfly species have high demands for habitat quality [including microclimate, vegetation structure, co-occurrence of vegetation types at a local scale - Thomas 1994] and they often respond quickly to habitat deterioration [e.g., 3 to 30 times faster than their host plants - Woiwod & Thomas 1993]. They are, therefore, generally considered to be useful indicators of habitat quality changes in particular terrestrial habitats [e.g., grasslands [Erhardt & Thomas 1991]] and have some potential to be an effective 'umbrella group' for biodiversity conservation [New 1997], although no single species or taxonomic group can be regarded as a universal bio-indicator [Simberloff 1998]. Nevertheless, the availability of distribution data since more than a century, and their specific relations with aspects of habitat quality, make butterflies a relevant group for analysing faunal changes in relation to changes in land-use.

In this paper, we analyse changes in butterfly diversity during the 20th century [and particularly during recent decades] in Flanders, the northern region of Belgium. This NW-European region is characterised by very high human population density and a high mean standard of living, resulting in high pressure on the environment [OECD 1998]. Natural habitats became human-dominated habitats several centuries ago. Although many different faunal elements went extinct during this process [e.g., large herbivores and large mammal predators, but probably also more inconspicuous organisms], many others were able to survive successfully in the traditionally-managed semi-natural habitats such as dry and wet heathlands, hayfields or coppiced woodlands. Several butterfly species and other thermophilous insects with relatively short-generation times [e.g., carabid beetles, grasshoppers] even became dependent on these early-successional habitat types for their survival [Thomas 1993]. Since about 1950, land-use progressively became more intensive [industry, agriculture, road and house building] and affected the landscape to a much greater extent. As a result, most traditionally-managed habitats were lost and present-day remnants are small and highly fragmented. Compared to other European countries or regions, nature reserves in north Belgium are very small: only 30 are larger than 100 ha and all reserves together occupy only 1.1% of the total area of Flanders [Decleer & De Belder 1999]. In the countryside the traditional landscape matrix [with small-scaled managed meadows, extended hedgerows, woodlands and large heathlands], is largely replaced by intensively used arable fields and sown grasslands [agro-industry]. From the 1960's, this process was accelerated by large agricultural land consolidation projects. Furthermore, there has been a diffuse spread of house building and an expansion of industrial zones [Table 3.1].

Table 3.1. Change of land use [in ha] in Flanders between 1834 and 1995. The figures for 1834 and 1980 are based on Van Der Haegen [1982], the figures for 1995 are based on Ministerie van de Vlaamse Gemeenschap [1996]. Between brackets the relative change [in percentage] between the area occupied in 1834 and that in 1995.

Land use	1834	1980	1995 [%change]
Built-on areas	11,670	135,120	202,239 [+1633%]
Other open space [e.g., dumping grounds, airfields, mine waste heaps…]	4,130	31,605	35,686 [+764%]
Gardens, parks and recreation zones	3,553	28,080	24,960 [+603%]
Roads, rivers and canals	44,985	93,046	106,745 [+137%]
Agricultural grasslands	169,230	311,670	296,815 [+75%]
Woodland	142,490	111,590	108,795 [-24%]
Arable land, horticulture, orchards	808,640	589,030	529,184 [-34%]
Heathlands and waste lands [= nutrient poor grasslands]	163,359	52,151	47,972 [-71%]

It is expected that such dramatic changes in land-use have a severe impact on the butterfly fauna [among other components of biodiversity] in Flanders. Here, we deal with changes in species numbers, changes in the extent of distribution and changes in hotspots with respect to [Red list] species richness. Since species differ in mobility and habitat use, and hence in their ability to survive in a changing and highly fragmented landscape, patterns are likely to differ among species. Turin & den Boer [1988] have shown that carabid beetles with poor dispersal abilities have declined more than those with good dispersal capacities. We may expect a similar pattern for butterflies with sedentary species showing stronger decreases in distribution area than the more mobile ones. Furthermore, the increased use of fertilisers since the 1950's and the over-production of manure by an over-sized stock of cattle [1,7 million – Lauwers *et al.* 1996] and pigs [6,8 million - Lauwers *et al.* 1996] have probably led to a stronger decrease in distribution area for species that are confined to oligotrophic habitats [e.g., nutrient poor grasslands and hayfields or heathlands] compared to species that thrive in eutrophic biotopes [e.g., abandoned meadows] [León-Cortés *et al.* 1999; Van Es *et al.* 1999; van Swaay 1990]. We therefore also examine how changes in distribution relate to mobility and to the habitat type of butterflies in Flanders.

To quantify and analyse changes in butterfly diversity, we used an extensive data set on the former and present-day distribution of the 64 indigenous species that was initially compiled for a documented distribution atlas [Maes & Van Dyck 1999]. Despite their great value and potential use for nature conservation, such distribution data inevitably carry several biases [e.g., temporal and spatial differences in recording effort] causing typical difficulties for the analyses [Dennis & Hardy 1999; Dennis *et al.* 1999; Dennis & Thomas 2000]. To reduce such effects maximally, a sub-data set fulfilling several criteria regarding recording intensity was used, rather than the entire data set.

Methods

Study area

Flanders [total area 1 351 200 ha] is the northern, Dutch speaking part of Belgium. It exhibits the typical features of a western industrialised region: extensive industry, infrastructure, house building and agriculture, and a human population density of 431 citizens/km^2 [Van Hecke & Dickens 1994]. The general landscape and topography differ considerably between Flanders and the southern part of Belgium [Wallonia]. Moreover, nature conservation policy is the responsibility of the regional governments rather than the Belgian federal government. Therefore, data from Wallonia were not incorporated. A survey of the butterflies of Wallonia, including a Red List, is given by Goffart *et al.* [1992]; van Swaay & Warren [1999] give the threat status of all indigenous butterflies in Belgium.

Origin of the data

For the Flemish butterfly atlas, about 190 000 records were collected on all butterfly species observed in Flanders since 1830 [Maes & Van Dyck 1999]. This extensive mapping scheme was co-ordinated by the Flemish Butterfly Working Group. Data came from: [1] collections of scientific institutions and private collectors [± 10 000 records], [2] reports in national and local journals [± 5 000 records] and [3] field observations [± 175 000 records]. Field observations were made by about 600 volunteers. Data from collections and from literature reports mainly dated from 1901-1980 while the field observations were mainly from 1985-2000. For all records at least the species, the year of observation and the exact location were noted. For all locations at least the 5x5 km Universal Transverse Mercator [UTM] square code was recorded and if possible even the 1x1 km UTM square code. These 5x5 km squares [*n*=644] were used as units of distribution. In our study we used the year 1991 as pivotal date because this coincides with the start of the large-scale butterfly-mapping project. A recent pivotal date reflects the current situation without possible time lags [species that have gone extinct in the meantime]. Fig. 3.1 gives an overview of the pre- and post-1991 coverage of the mapping scheme with numerical values for species richness per square and Fig. 3.2 shows the number of records per five-year period in the 20th century.

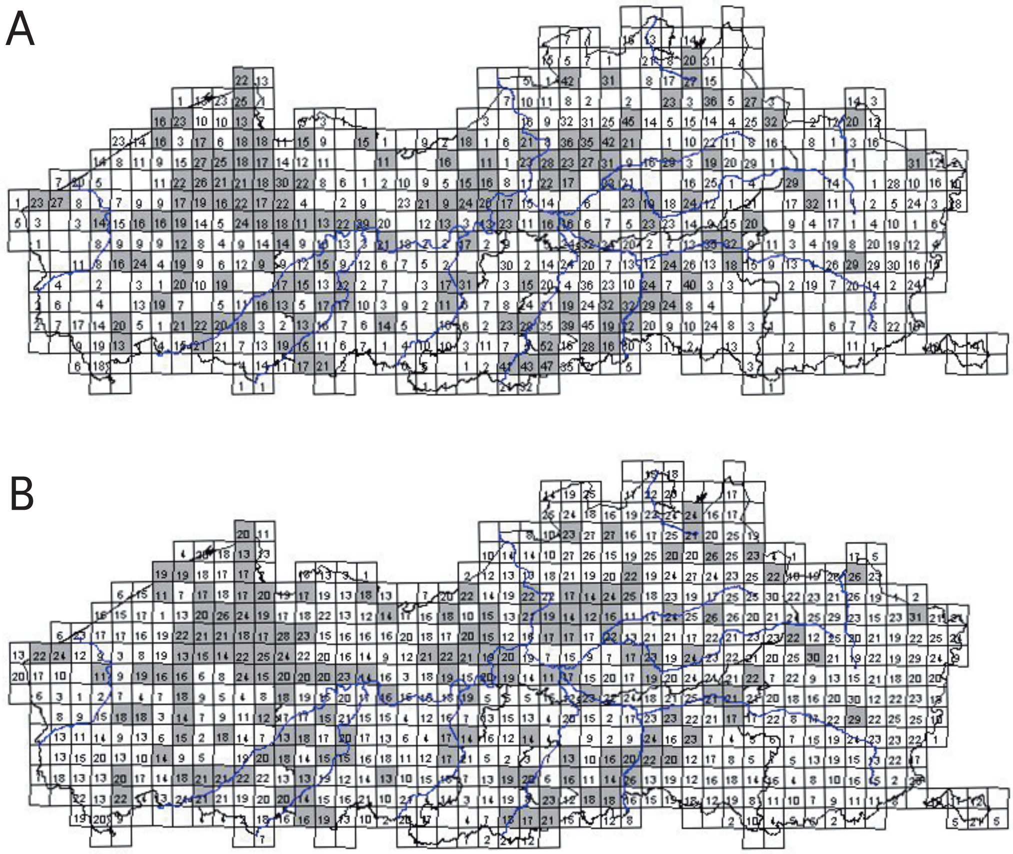

Figure 3.1. Number of species per square before 1991 [a] and since 1991 [b]; squares used in the analyses are shaded in grey.

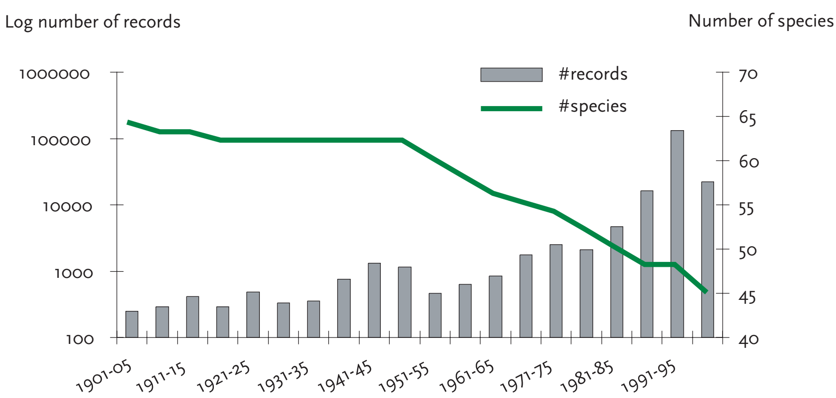

Figure 3.2. Number of records [left Y-axis, log scale] and number of species [right Y-axis] per five year period in Flanders in the 20th century.

Changes in species numbers and in diversity and Red List species hotspots

To determine changes in species numbers, we counted the number of indigenous species per five-year period during the 20th century. This enabled us to calculate an extinction rate for butterflies [Thomas & Morris 1994] and to compare the present species richness with the historical one.

We determined present and historical species rich areas ['hotspots'] by counting the number of species per square before and after 1991. Both the total number of species per square [i.e., diversity hotspots, DHS] and the number of Red List species per square [i.e., Red List species hotspots, RLHS] were analysed. Each species was assigned a Red List status according to Maes & Van Dyck [1999; Appendix 3.1]. Red List categories are those proposed by the IUCN Species Survival Commission [1994], adapted to Flanders [Maes *et al.* 1995; Maes & van Swaay 1997]. DHS and RLHS are arbitrarily defined as the top 5% of the recorded squares, ranked by decreasing number of all species and of Red List species respectively, in the period after 1991 [Prendergast *et al.* 1993b]. DHS in Flanders are determined as squares with ≥ 26 species and RLHS as squares with ≥ five Red List species. To estimate changes in the number of DHS and RLHS between the two periods, we used criteria similar to Prendergast & Eversham [1995]: hotspots are determined using the present-day data, and changes in the numbers of hotspots are relative to the recent period [i.e., since 1991].

Changes in distribution area

To analyse changes in distribution area, we compared the number of squares in which each species was recorded before and after 1991. However, because the data originate from different sources [collections, literature citations and field observations] and were collected on a voluntary basis, there is a bias in sampling effort both in time, space and targeted species [Dennis & Hardy 1999; Dennis *et al.* 1999; Dennis & Thomas

2000]. We therefore restricted the analyses to squares we considered sufficiently well investigated in both periods. To select these squares, we counted the number of UTM squares in which each species was mapped during both periods and ranked them in decreasing number of squares. For both periods the top six consisted of the same species [although in a different order], i.e. *Pieris brassicae*, *Pieris rapae*, *Pieris napi*, *Inachis io*, *Aglais urticae* and *Pararge aegeria*. We restricted the analysis to those squares in which all these six species were recorded in both periods. This criterion restricted the analysis to 150 squares [23% of all squares]. Seven species were excluded from the analysis because they reach the margin of their distribution area in Flanders [according to Tolman & Lewington [1997] and Bink [1992]; Appendix 3.1]. This reduced the number of species analysed to 57 of the 64 indigenous species. The 150 squares used in this analysis are fairly well spread over the geographical regions of Flanders [Dufrene & Legendre 1991]: 33 squares are situated in the Loamy region [17% of all squares in this region], 34 squares in the Campine region [17% of all the squares in this region], 13 squares in the Coastal region [19% of all the squares in this region] and 70 squares in the Sandy-loamy region [37% of all the squares in this region]. Therefore, we consider the 150 squares used for the analyses as representative of Flanders. Furthermore, the number of records is comparable in both periods: 12 393 records [i.e., a species in a square in a given year] date from before 1991 and 10 035 records date from since 1991.

Changes in distribution area in relation to mobility and habitat type

To analyse relationships between changes in distribution area on the one hand, and dispersal capacity and habitat type on the other hand, all indigenous species were assigned to a mobility class and to a nutritional value of the breeding habitat [Appendix 3.1]. Mobility for each species was derived using Bink's [1992] method of nine mobility classes, ranging from very sedentary species to wanderers. To avoid small sample sizes, the number of classes was reduced to four [1 = very sedentary, 2 = sedentary, 3 = fairly sedentary, 4 = mobile-very mobile] by lumping the first and second on the one hand, and the fifth, sixth and seventh mobility classes on the other hand; the eight and ninth mobility class refer to vagrants that are not indigenous in Flanders [e.g., *Vanessa cardui*]. For the nutritional value of the breeding habitat, the classification is based on the average nutrient number [Stickstoffzahl, Ellenberg *et al.* 1992] of the hostplant[s] [Oostermeijer & van Swaay 1998] and additional ecological literature [e.g., Bink 1992; Emmet & Heath 1989; Tax 1989]. We distinguished three nutritional values of the breeding habitat: [1] oligotrophic, [2] mesotrophic and [3] eutrophic. To avoid a tendency towards no change in distribution area of the most common species, the six species used to determine the sub-set of squares [*Pieris brassicae*, *Pieris rapae, Pieris napi*, *Inachis io*, *Aglais urticae* and *Pararge aegeria*,] were excluded from this analysis. This reduced the number of species analysed from 57 [see above] to 51.

Statistical analysis

The complete data set was used to determine changes in species numbers and for the hotspot analysis. Changes in distribution area in relation to mobility and to nutritional value of the breeding habitat were analysed with the sub-data set as described above. The changes in distribution area were defined as "Log_{10} [number of squares since 1991+1] - Log_{10} [number of squares before 1991+1]" [Appendix 3.1]. Effects of mobility and nutritional value of the breeding habitat on changes in distribution area were tested separately by one-way analysis of variance since mobility and breeding habitat were not statistically independent [Sokal & Rohlf 1995].

Results

Butterfly diversity and Red List species

Butterfly species richness decreased strongly during the 20th century: 44 species show a negative trend [of which 21 declined significantly, χ^2-test: p<0.05], while only 13 species show a positive trend [of which 4 increased significantly, χ^2-test: p<0.05]. Seven species show no trend: the six species used to determine the restricted data set and *Heteropterus morpheus*, a very rare species restricted to one grid square. The number of species that decreased or increased significantly and remained stable differs highly significant from what can be expected under the null-hypothesis of no changes [χ^2=28.7, p<0.001]. Of the 35 species that were considered very rare to fairly common before 1991 [present in 1-50 squares, i.e., 1/3 of the total number of squares in the sub-data set], all but three [*Thymelicus sylvestris*, *Aricia agestis* and *Melanargia galathea*] decreased in distribution area. On the other hand, 11 of the 16 species that were common to very common before 1991 [present in 51-150 squares] increased in distribution area; two significant exceptions were *Coenonympha pamphilus* and *Lasiommata megera*.

The number of species per five-year period decreased since the beginning of the century from 64 in 1901-1905 to 45 in 1996-2000 [Fig. 3.2]. This is an average extinction rate [Thomas & Morris 1994] of 0.95 species per five year period during the 20th century. However, if we compare the first and second half of the century [1901-1950 vs. 1951-2000], the extinction rate was 0.20 species/five year period and 1.70 species/five year period respectively which means that it increased more than eightfold during the second half of the 20th century.

Diversity hotspots [DHS] and Red List hotspots [RLHS]

In the period before 1991, 57 squares were determined as diversity hotspots [DHS] of which 51 fell below the threshold of 26 species in the period after 1991. Only six squares preserved their status as DHS. In the period after

Figure 3.3. Lost [crosses], maintained [black dots] and gained [grey dots] diversity hotspots [top] and Red List species hotspots [bottom] in Flanders.

1991, 16 squares were gained as DHS compared to the period before 1991 [Fig. 3.3].

In the period before 1991, 107 squares were determined as Red List hotspots [RLHS] of which 96 fell below the threshold of five Red List species in the period after 1991. Only 11 preserved their status as RLHS. In the period after 1991, 14 squares were gained as RLHS compared to the period before 1991 [Fig. 3.3]. A higher [spatial] recording intensity in the second period is most likely responsible for the 'gained' DHS and RLHS [Table

Table 3.2. Average number of species in the lost, maintained and gained diversity hotspots [DHS] before and since 1991 and average number of Red List species in the lost, maintained and gained Red List hotspots [RLHS] before and since 1991. Between brackets the average number of records per square per five year period in lost, maintained and gained hotspots before and since 1991.

	DHS		RLHS	
	<1991	≥1991	<1991	≥1991
Lost hotspots	32.94 [5.8]	18.71 [72.7]	9.88 [4.9]	1.97 [61.4]
Maintained hotspots	31.50 [13.0]	28.50 [187.1]	9.00 [9.1]	6.18 [141.9]
Gained hotspots	14.57 [10.1]	27.19 [150.0]	1.64 [8.8]	5.79 [111.0]

3.2] and not the colonisation of new squares by [Red List] species. The average number of [Red List] species and records in the former and present DHS and RLHS is given in Table 3.2.

Hotspots [both DHS and RLHS] are mainly lost around Brussels in the south, around Antwerp in the north and in the dune area [Fig. 3.3]. With very few exceptions, the present-day hotspots are situated on the sandy soils in NE-Flanders [Campine region] where heathlands, nutrient poor grasslands and woodlands still co-occur.

Changes in distribution area in relation to mobility and habitat type

Changes in distribution area differed significantly with level of mobility and with nutritional value of the breeding habitat. Butterfly species with a low dispersal capacity experienced a more severe loss in distribution area than species with a higher dispersal capacity: Kruskal-Wallis ANOVA: H [3, N=51]=10.187, p=0.017 [Fig. 3.4]. Species of oligotrophic habitats experienced a more severe loss in distribution area than did species of mesotrophic and eutrophic habitats: Kruskal-Wallis ANOVA: H [2, N=51]=15.781, p<0.001 [Fig. 3.4]. Posteriori tests for differences in changes in distribution area among mobility classes and among nutrient values of the breeding habitat are given in Fig. 3.4.

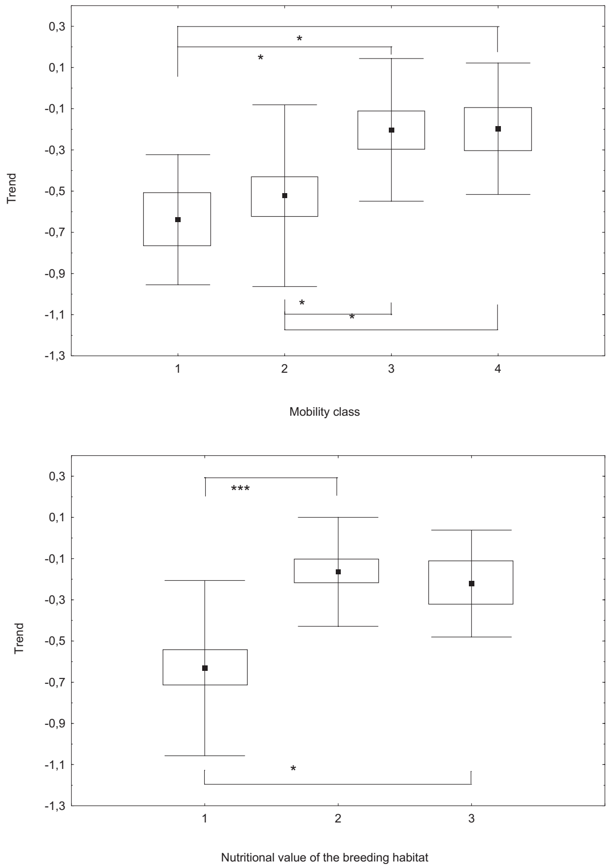

Figure 3.4. Mean trend, standard error and standard deviation per mobility class [top] and nutritional value of the breeding habitat [bottom]. Number of species per mobility class [MC] is: MC 1 [n=6], MC 2 [n=22], MC 3 [n=14], MC 4 [n=9]. Number of species per nutrient value of the breeding habitat [NV] is: NV 1 [n=24], NV 2 [n=21], NV 3 [n=6]. A posteriori tests [Least significant difference test] for differences in changes in distribution area [a] among mobility classes and [b] among nutrient values of the breeding habitat are indicated by lines between classes [only significant differences are shown]: * = p<0.05; *** = p<0.001.

Discussion

Biases in comparing historical and recent data

Establishing the conservation status and distribution trends of species – and hence of at least parts of the biodiversity – for a political relevant unit [region or country], has been a traditional and valuable tool to evaluate the efficiency of nature conservation efforts. However, when comparing former and present-day distribution data, biases in time, space and targeted species [rare vs. common species] can hardly ever be excluded due to differences in collection methods and time periods considered [Dennis *et al.* 1999; Dennis & Thomas 2000]. Restricting the comparison to records from butterfly collections was not possible since hardly any collection data are available for the recent time period [since 1991]. We have attempted to maximally reduce the biases in the data set by using a limited set of well investigated squares during the two time periods. Other criteria to obtain subsets of well investigated squares [e.g., at least 15 species present/square, at least 75% of the very common species present/square] or a different pivotal date [e.g., 1980, 1970 or even 1950], yielded very similar results as the ones described in this paper, and do not change the main conclusion that Flanders is one of the regions with the severest butterfly loss in Europe. Bias is mainly due to the undermapping of most common species in the earlier periods compared to recent butterfly mapping schemes. Rare species however, were fairly well mapped both before and after 1991. Before 1991 rare species were actively looked for and therefore well represented in collections and in literature citations; after 1991 the Flemish butterfly mapping scheme placed an emphasis on detailed mapping of the distribution of Red List species in detail on topographic maps. Declining trends of rare species are, although severe for most of them, probably still underestimated since they are based on relatively coarse-grained distribution data

[Thomas & Abery 1995] and since not all past populations were mapped before they went extinct, whereas almost all recent populations of rare butterflies are known. On the basis of UTM squares most common species seem to have a stable or even an increasing distribution, but at the population level, these species may show an equally strong decrease as some of the rare species [Cowley *et al.* 1999; León-Cortés *et al.* 1999, 2000]. Declining relative abundances of still widely distributed butterfly species [e.g., *Inachio io*, *Aglais urticae*] have been noticed from butterfly transect counts in the 1990's in Flanders and The Netherlands [van Swaay & Ketelaar 2000; van Swaay *et al.* 1997].

Butterfly diversity loss

In Flanders 30% of the indigenous butterfly species went extinct during the 20th century and half of the remaining species is threatened [Maes & Van Dyck 1999]. Furthermore, about 90% of the former hotspots [both diversity and Red List species hotspots] have been lost despite a strong increase in recording intensity [Table 3.2 and Fig. 3.2]. The strong deterioration of the butterfly fauna is not restricted to Flanders, but also affects most other NW-European countries or regions: the Netherlands [Wynhoff & van Swaay 1995], Wallonia [S-Belgium] [Goffart & De Bast 2000], Baden-Württemberg [Ebert & Rennwald 1993b] and Germany [Pretscher 1998] [Table 3.3]. Denmark [van Swaay & Warren 1999] and Great Britain [Warren *et al.* 1997] are the only two NW-European countries with a relatively limited number of extinct and threatened species. Within Flanders, butterflies also show the highest number of extinct and Red List species compared to other taxonomic groups for which the conservation statuses are known [Table 3.4].

Table 3.3. Comparison of the number of extinct and Red List species among some NW-European countries or regions. The countries or regions are ordered in decreasing percentage of Extinct and Red List [%E+RL] species.

Country or region	Extinct	Red List	%E+RL	Total number
Flanders [Maes & Van Dyck 1999]	19 [30%]	22 [34%]	64%	64
The Netherlands [Wynhoff & van Swaay 1995]	17 [24%]	28 [40%]	64%	70
Wallonia [Goffart 2000]	15 [14%]	50 [48%]	63%	104
Baden-Württemberg [Ebert & Rennwald 1993b]	4 [3%]	80 [58%]	61%	137
Germany [Pretscher 1998]	6 [3%]	87 [47%]	50%	185
Denmark [van Swaay & Warren 1999]	4 [6%]	18 [26%]	35%	68
Great-Britain [Warren *et al.* 1997]	4 [7%]	8 [14%]	20%	59

Table 3.4. Comparison of the number of Red List species among different taxa in Flanders. The taxonomic groups are ranked in decreasing percentage of Extinct and Red List species [%E+RL].

Taxonomic group	Extinct species	Red List species	%E+RL	Total number
Butterflies [Maes & Van Dyck 1999]	19 [30%]	22 [34%]	64%	64
Dragonflies [De Knijf & Anselin 1996]	9 [16%]	20 [34%]	50%	58
Amphibians/reptiles [Bauwens & Claus 1996]	2 [11%]	6 [32%]	42%	19
Mammals [Criel 1994]	11 [18%]	13 [22%]	40%	60
Spiders [Maelfait *et al.* 1998]	9 [1%]	204 [34%]	35%	604
Empidid flies [Grootaert *et al.* 2001]	31 [13%]	49 [20%]	32%	248
Vascular plants [Cosyns *et al.* 1994]	81 [6%]	325 [25%]	32%	1288
Breeding birds [Devos and Anselin, in prep.]	4 [3%]	44 [28%]	30%	159
Carabid beetles [Desender *et al.* 1995]	32 [9%]	66 [19%]	28%	352
Fish [Vandelannoote *et al.* 1998]	11 [20%]	2 [4%]	24%	55
Dolichopodid flies [Pollet 2000]	22 [8%]	39 [15%]	23%	260

The decline in the number of hotspots and the regions where they were lost are largely coincident for overall species richness and Red List species richness. For instance, the larger woodlands around Brussels [e.g., the Walenboscomplex - Tips 1977] used to hold many butterfly species typical of open woodland, grasslands, and even heathlands on sandy areas in the woodlands. Almost all these butterfly species disappeared from these situations because of, either economic exploitation of the woodlands, or a lack of appropriate conservation management. In the dune area, the strong expansion of house building for tourism considerably reduced the area of semi-natural grasslands; ceasing of grazing in several remnants [Vermeersch 1986] further reduced the availability of early-successional habitats for several Red List species. In NE-Flanders, the majority of heathlands and nutrient-poor grasslands were transformed into arable land, conifer plantations or other land-uses, although some small to medium-sized heathlands and deciduous woodlands still remain, often in military areas.

The extinction rate of butterfly species in Flanders has increased markedly compared to the first part of the 20th century, despite an increasing total number of data since 1950 [Fig. 3.2]. Several factors that contributed to this decline are still operating. Indeed, the Flemish government has recently received a European Commission' s letter giving formal notice [i.e., the first step in the procedure towards a conviction] for insufficient efforts to reach the minimal standards regarding nitrogen pollution due to the over-production of manure [EU Nitrogen Directive] and one regarding the completion of the habitat network NATURA 2000 [EU Habitat Directive]. Assuming a linear extinction rate of 1.70 species/five year [as in the second half of the 20th century], Flanders will lose the 22 remaining Red List species in a period of only 65 years.

As in most other NW-European countries, destruction of suitable butterfly habitats, habitat fragmentation and declining habitat quality are the factors responsible for the deterioration of the butterfly fauna [Goffart *et al.* 1992; Maes & Van Dyck 1999; Thomas 1984, Thomas 1991; van Swaay 1990; Warren *et al.* 1997]. In Belgium and in the Netherlands in general and in Flanders in particular, the increase in built-on area and the intensification and expansion of agriculture [Table 3.1] destroyed, or decreased the quality of, many suitable butterfly habitats. Species have disappeared even within nature reserves or other protected areas and still are disappearing due to inappropriate management [Thomas 1984; van Swaay 1990; Warren 1993]. Nature management has often been focused on vascular plants and particularly birds which is not necessarily suitable for butterflies and other invertebrates [Thomas 1994]. Two recent cases illustrate how inappropriate management caused great damage and even the extinction of highly threatened butterfly populations. In a Flemish nature reserve overgrazing of the host plant *Gentiana pneumonanthe* lead to the local extinction of *Maculinea alcon*, while in the Netherlands, a wrong mowing date severely reduced the number of individuals in the reintroduced populations of *Maculinea teleius* [Wynhoff 1998a].

Of the species with the strongest decline in distribution area, all but two used to be relatively rare in the past, while of the species that increased their distribution area all but two were already widespread in the past. A similar pattern has been shown for butterflies in the UK [Pollard & Yates 1993] and for amphibians and reptiles in Flanders [Bauwens & Claus 1996]. The extinction of a local population of a rare butterfly is therefore often definitive, while common and more mobile species are able to colonise new sites.

Changes in distribution area in relation to mobility and habitat type

Species with limited dispersal capacities and species restricted to oligotrophic habitats decreased significantly more strongly than mobile species and than species from eutrophic habitats. The fact that less mobile species decreased more strongly than the more mobile ones seems obvious. Once the habitat of a sedentary species is destroyed, it is unable to find suitable habitat patches that are beyond its dispersal range. Furthermore, the loss of a habitat patch has far greater consequences for a sedentary species in a metapopulation than for a mobile one in an open population: metapopulations of sedentary species fall apart in isolated populations and become more susceptible to extinction [Harrison 1991; Thomas & Hanski 1999]. At present, habitat patches are destroyed at a much higher speed than the colonisation rate of most species living in a metapopulation structure [e.g., *Melitaea cinxia* is now confined to some canal borders and road verges in the NE of Flanders].

Species of oligotrophic habitats used to occur in the traditionally-managed agricultural landscape. Since agriculture became more intensive in the second half of the 20th century [e.g., artificial fertilisers, increasing numbers of stock on farms, land consolidation projects - Nysten 1994], many [thermophilous] butterfly species were no longer capable of completing their life cycle due to an increased mowing frequency in intensively used agricultural land or due to taller vegetation, and thus a colder microclimate, in abandoned hayfields [Thomas 1993]. Belgium is one of the countries in Europe with the highest nitrogen input on arable land [17.6 tons/km^2 arable land - OECD 1998]. Furthermore, since agriculture is much more intensive in Flanders than in Wallonia, this figure is probably an underestimate for the northern part of the country. On average, arable land in Flanders has a surplus on the nutrient balance of 236 kg N/ha and of 34 kg P/ha [Vanongeval *et al*. 1998]. The excessive use of nitrogen and phosphorus has direct consequences on the vegetation structure of grasslands and hence on grassland butterflies [Geypens *et al*. 1994; Thomas 1993]. Indirectly, nutrient deposition and groundwater polluted with nutrients can change the vegetation type and structure in wet heathlands [causing domination of *Molinia* *caerulea*], in woodlands [causing domination of *Deschampsia flexuosa* and declining tree quality] and in marshes and moist grasslands [Geypens *et al.* 1994]. Most parts of Flanders have a nitrogen deposition of less than 30 kg/ha/year, but in intensive agricultural areas it reaches peaks of more than 50 kg/ha/year [in N-Flanders] to even 70 kg/ha/year [in W-Flanders] [Vanongeval *et al.* 1998]. Examples of species of oligotrophic grasslands showing extreme declines are *Lycaena tityrus* [Fig. 3.5, see also van Swaay 1995], *Polyommatus semiargus* and *Melitaea cinxia* in dry habitats and *Boloria selene* in wet habitats. Even fairly common species of dry grassland habitats [e.g., *Lycaena phlaeas*, *Polyommatus icarus*, *Coenonympha pamphilus*, and *Lasiommata megera*] show declining trends that are most probably still underestimated from a census with a coarse grained grid [Cowley *et al.* 1999; León-Cortés *et al.* 1999, 2000].

A

B

C

D

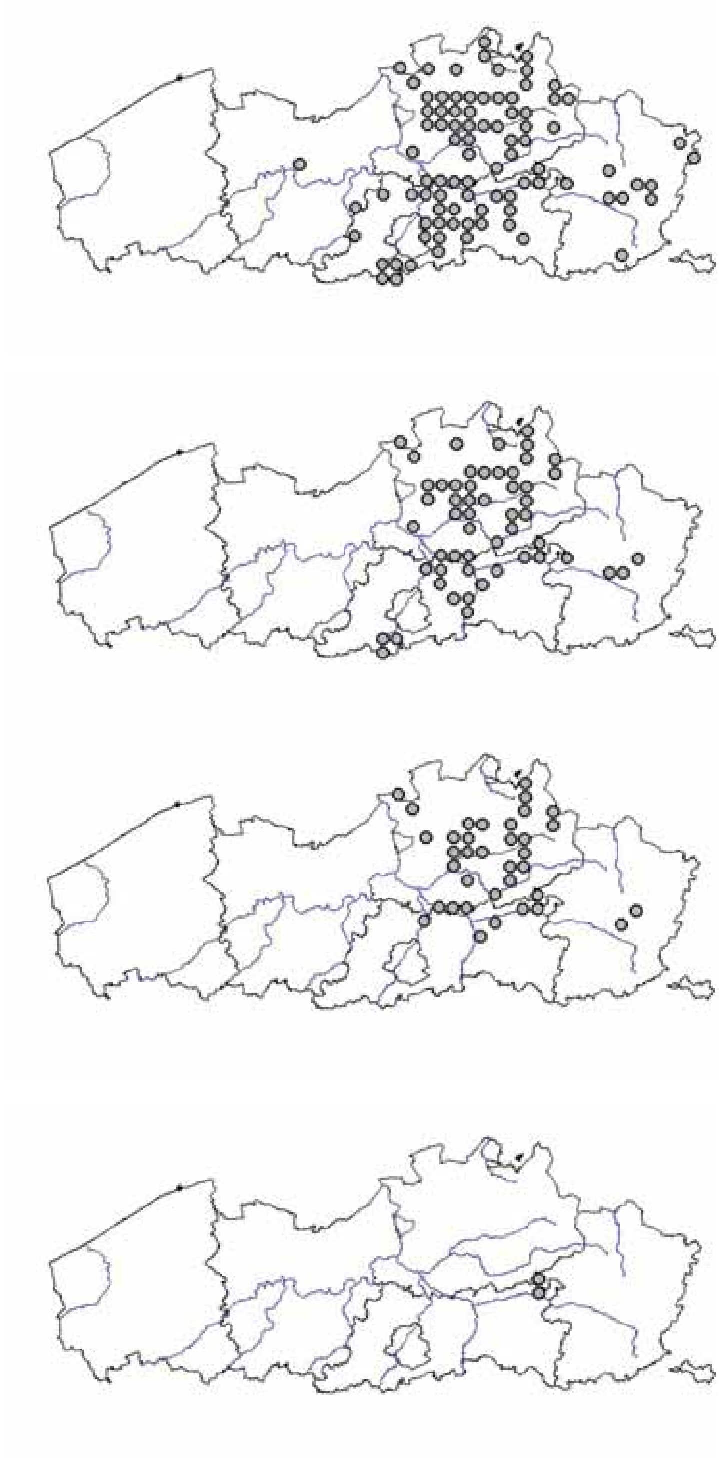

Figure 3.5. Distribution of *Lycaena tityrus* in Flanders in the 20th century: a. 1901-1950, b. 1951-1970, c. 1971-1990, d. 1991-2000.

Butterfly and biodiversity conservation in Flanders

Given the large number of extinct and threatened butterfly species, particularly from habitats with a high conservation value [e.g., dry and wet heathlands, open broad-leaved woodland, flower-rich hay meadows], Flanders urgently needs extra efforts regarding the conservation and restoration of both the quantity and quality of habitat networks. The Flemish government and Ministry of Nature Conservation are well aware of this need and since 1997 a new Decree on Nature Conservation offers much more opportunities and tools to do so. However, the new approaches to realise the Flemish Ecological Network [Decleer *et al.* 1999] including several new nature development and consolidation projects, are largely site-oriented. Hence, biological realism may be lacking when crucial features of target species [e.g., dispersal] have not been taken into account [e.g., efficiency of corridors, metapopulation structure - Baguette *et al.* 2000], etc.]. Moreover, the general configuration of ecological networks in Flanders will mainly be affected by non-scientific decision rules [e.g., budgets, agreements with other land-users, etc.]. Scientific evaluation of different scenarios for the completion of these networks seems crucial in order to maximise the effect of conservation efforts and budgets on the preservation and restoration of biodiversity. Among other taxonomic groups, butterflies may play a valuable role for planning and evaluating site-oriented conservation measures. However, it will be essential to develop Biodiversity Action Plans for each threatened species.

A large number of populations of threatened species are nowadays restricted to nature reserves or to large military areas. Many of these reserves are too small to contain sustainable populations and undergo strong negative influences from outside the reserve [e.g., nitrogen deposition, recreation from nearby cities, etc.]. Priority should be given to the enlargement of existing reserves [including appropriate management – for reviews see Gibbons *et al.* 1993; New *et al.* 1995; Smallidge & Leopold 1997] and to the acquisition of non-protected sites with threatened butterflies, especially in the NE of Flanders where the [Red List] species richness is relatively least impoverished [Fig. 3.3]. The new Flemish Ecological Network is aiming to create an ecological network of [1] large nature units (total area 125, 000 ha], [2] large nature development units [total area 150, 000 ha] and [3] ecological corridors. However, populations of some Red List species [*Thecla betulae, Satyrium w-album, Cupido minimus, Aricia agestis, Polyommatus semiargus and Melitaea cinxia*] are mainly situated outside the preliminary defined networks and need additional protection measures. The completion of ecological corridors is also questionable since no particular species have yet been used as role models. However, little is known about the use of particular corridors or 'stepping stones' by butterflies and other invertebrates and further research on the requirements of corridors is urgently needed [Haddad 1999; Haddad & Baum 1999]. Nevertheless, a critical screening on the basis of best available current knowledge would be highly valuable. For several heathland species large military areas are the strongholds in Flanders [e.g., *Maculinea alcon, Plebeius argus, Hesperia comma, Callophrys rubi* and *Hipparchia semele*]. The Flemish Ministry for nature conservation recently signed a protocol with the military authorities that obliges the latter to take into account the presence of threatened species and to draw up specific management plans for military areas.

Legal species protection [in particular, a ban on capturing and collecting] has shown to be a very inadequate measure to ensure their conservation in Flanders [only one [*Maculinea alcon*] of the 13 legally protected species is actually present in Flanders]. Furthermore, collecting is only harmful in very small and isolated populations [Thomas 1983]. Hence, a series of species action plans or multi-species action plans [for particular threatened habitats] would be more useful. Presently the first species action plan for an invertebrate is in preparation: i.e., *Maculinea alcon*, an endangered species in Flanders and Belgium [Maes & van Swaay 1997] and a vulnerable species in Europe [van Swaay & Warren 1999]. Based on national and international relevance, species action plans are desirable for *Pyrgus malvae, Hesperia comma, Issoria lathonia, Boloria selene, Melitaea cinxia and Coenonympha tullia*. The preservation of existing populations of threatened species remains the first priority; it is only in the second place that reintroduction should be considered [e.g., Ravenscroft 1992]. Potential species for local reintroductions [in order to enforce or recreate a metapopulation structure, cf. Thomas & Jones 1993] are *Pyrgus malvae, Hesperia comma, Maculinea alcon, Plebeius argus* and *Melitaea cinxia*. Research on the feasibility of national reintroductions of the following species for which potentially suitable habitats are still present in Flanders can be considered: *Maculinea teleius* [extinct since 1980], *Plebeius idas* [extinct since 1984], *Argynnis niobe* [extinct since 1977] and *Euphydryas aurinia* [extinct since 1959]. Since several authors have shown that some of the common butterflies are declining as severely as the threatened ones [Cowley *et al*. 1999; León-Cortés *et al*. 1999, 2000], additional measures [education, management advice, ...] should be taken to preserve the "common" butterflies in the agricultural and rural landscape.

In conclusion, the Flemish butterfly data set is, like most data sets containing historical and recent data, biased in both time and space and in targeted species [rare vs. common species]. However, by using a subset of data in which this bias has been reduced maximally, we were able to demonstrate that Flanders is one of the regions with the largest loss in butterfly diversity in Europe. Species with low dispersal abilities and species of oligotrophic habitats suffered most of habitat fragmentation and/or intensification of agriculture. Flanders therefore urgently needs to take actions and adjust current actions to preserve its endangered butterflies and their habitats.

Appendix 3.1. Resident butterflies in Flanders with the number of squares in which the species was observed before and since 1991 in a selection of well mapped squares [for explanation, see text]. Species names are according to Karsholt & Razowski [1996]. [w] = some of the squares may be observations of wanderers. Trend = Log10 [number of squares since 1991+1] – Log10 [number of squares before1991+1]; significant decreasing/increasing trends are indicated by --- [Fisher exact two tailed p<0.001], -- [Fisher exact two tailed p<0.01], -/+ [Fisher exact two tailed p<0.05]. RLC = Red List category: Ex = Extinct in Flanders; CE = Critically endangered, CE [Ex] = categorised as Critically endangered but extinct in the meantime; En = Endagered, Vu = Vulnerable, IK = Insufficiently known, R = Rare, S/LR = Safe/Low risk. Mobility class [based on Bink 1992]: 1 = very sedentary, 2 = sedentary, 3 = fairly sedentary, 4 = mobile-very mobile. Nutritional value of the breeding habitat [based on the average nutrient number of the hostplant[s] according to Ellenberg et al. [1992]: 1 = oligotrophic, 2 = mesotrophic, 3 = eutrophic.

Species	before 1991	since 1991	Trend	RLC	Mobility	Nutritional value
Erynnis tages	11	-	-1.079 [---]	Ex	2	1
Pyrgus malvae	23	4	-0.681 [---]	En	2	1
Carterocephalus palaemon	19	11	-0.222	Vu	2	2
Thymelicus lineola	99	118	+0.076 [+]	S/LR	3	2
Thymelicus sylvestris	52	59	+0.054	S/LR	2	2
Hesperia comma	19	5	-0.523 [--]	En	2	1
Ochlodes venata	99	107	+0.033	S/LR	3	2
Papilio machaon	89	97	+0.037	S/LR	4	2
Leptidea sinapis	10	5	-0.263	CE	3	2
Anthocharis cardamines	99	116	+0.068 [+]	S/LR	3	2
Aporia crataegi[w]	19	3	-0.699 [---]	Ex	4	2
Pieris brassicae	150	150	0	S/LR	4	3
Pieris rapae	150	150	0	S/LR	4	3
Pieris napi	150	150	0	S/LR	4	3
Gonepteryx rhamni	133	140	+0.022	S/LR	4	2
Lycaena phlaeas	138	128	-0.032	S/LR	3	1
Lycaena tityrus	37	2	-1.103 [---]	CE [Ex]	2	1
Thecla betulae	17	11	-0.176	En	2	2
Neozephyrus quercus	44	37	-0.073	S/LR	2	2
Callophrys rubi	27	17	-0.192	Vu	3	1
Satyrium w-album	13	2	-0.669 [--]	IK	1	3
Satyrium ilicis	29	9	-0.477 [--]	Vu	2	2
Celastrina argiolus	121	131	+0.034	S/LR	4	2
Maculinea teleius	1	-	-0.301	Ex	1	1

Species	before 1991	since 1991	Trend	RLC	Mobility	Nutritional value
Maculinea alcon	9	4	-0.301	En	1	1
Plebeius argus	29	8	-0.523 [---]	Vu	2	1
Plebeius idas	1	-	-0.301	Ex	2	1
Aricia agestis	32	35	+0.038	Vu	3	1
Polyommatus semiargus	24	1	-1.097 [---]	CE	3	1
Polyommatus icarus	130	128	-0.007	S/LR	3	1
Argynnis paphia[w]	19	11	-0.222	CE	3	2
Argynnis aglaja[w]	14	3	-0.574 [-]	Ex	2	1
Argynnis adippe	5	-	-0.845	Ex	3	2
Argynnis niobe	4	-	-0.699	Ex	2	1
Issoria lathonia[w]	37	6	-0.735 [---]	CE	4	1
Boloria euphrosyne	10	-	-1.041 [--]	Ex	2	1
Boloria selene	27	-	-1.447 [---]	CE [Ex]	2	1
Inachis io	150	150	0	S/LR	4	3
Aglais urticae	150	150	0	S/LR	4	3
Polygonia c-album	128	141	+0.042 [+]	S/LR	4	3
Araschnia levana	128	138	+0.032	S/LR	4	3
Nymphalis antiopa	18	10	-0.237	Ex	4	3
Nymphalis polychloros	40	21	-0.270 [--]	En	4	3
Euphydryas aurinia	12	-	-1.114 [---]	Ex	1	1
Melitaea cinxia	21	1	-1.041 [---]	CE	2	1
Melitaea diamina	3	-	-0.602	Ex	1	2
Melitaea athalia	10	-	-1.041 [--]	Ex	2	1
Limenitis camilla	26	16	-0.201	Vu	2	2
Apatura iris	9	5	-0.222	En	2	3
Pararge aegeria	150	150	0	S/LR	3	2
Lasiommata megera	127	100	-0.103 [---]	S/LR	3	2
Coenonympha tullia	6	-	-0.845 [-]	CE [Ex]	1	1
Coenonympha pamphilus	128	94	-0.133 [---]	S/LR	2	1
Pyronia tithonus	87	105	+0.081 [+]	S/LR	2	2
Aphantopus hyperantus	78	68	-0.059	S/LR	2	2
Maniola jurtina	121	127	+0.021	S/LR	3	2
Hipparchia semele	47	19	-0.380 [---]	Vu	3	1
Species at the edge of their distribution area in Flanders [not used in the analysis]						
Pyrgus armoricanus	2	-	-0.477	Ex	2	1
Heteropterus morpheus	1	1	0	R	1	2
Cupido minimus[w]	2	1	-0.176	R	1	1
Limenitis populi	3	-	-0.602	Ex	2	2
Coenonympha hero	3	-	-0.602	Ex	1	2
Melanargia galathea[w]	4	7	+0.204	R	2	1
Hipparchia statilinus	3	-	-0.778	Ex	2	

4 Modelling as a pro-active tool in nature conservation

foto: Valérie Goethals

We evaluate differences between and the applicability of three linear predictive models to determine butterfly hotspots in Belgium for nature conservation purposes. The study is carried out in Belgium for records located to Universal Transverse Mercator [UTM] grid cells of 5x5 km. We first determine the relationship between factors correlated to butterfly diversity by means of modified t-tests and principal components analysis; subsequently, we predict hotspots using linear models based on land use, climate and topographical variables of well-surveyed UTM grid cells [N=197]. The well-surveyed squares are divided into a *training set* and an *evaluation set* to test the model predictions. We apply three different models: 1] a 'statistically-focused' model where variables are entered in descending order of statistical significance, 2] a 'land use-focused' model where land use variables known to be related to butterfly diversity are forced into the model and 3] a 'hybrid' model where the variables of the 'land use-focused model' are entered first and subsequently complemented by the remaining variables entered in descending order of statistical significance. A principal components analysis reveals that climate, and to a large extent, land use are locked into topography, and that topography and climate are the variables most strongly correlated with butterfly diversity in Belgium. In the statistically-focused model, biogeographic region alone explains 65% of the variability; other variables entering the statistically-focused model are the area of coniferous and deciduous woodland, elevation and the number of frost days; the statistically-focused model explains 77% of the variability in the training set and 66% in the evaluation set. In the land use-focused model, biogeographic region, deciduous and mixed woodland, natural grassland, heathland and bog, woodland edge, urban and agricultural area and biotope diversity are forced into the model; the land use-focused model explains 68% of the variability in the training set and 57% in the evaluation set. In the hybrid model, all variables from the land use-focused model are entered first and the covariates elevation, number of frost days and natural grassland area are added on statistical grounds; the hybrid model explains 78% of the variability in the training set and 67% in the evaluation set. Applying the different models to determine butterfly diversity hotspots resulted in the delimitation of spatially different areas. The best predictions of butterfly diversity in Belgium are obtained by the hybrid model in which land use variables relevant to butterfly richness are entered first after which climatic and topographic variables were added on strictly statistical grounds. The land use-focused model does not predict butterfly diversity in a satisfactory manner. When using predictive models to determine butterfly diversity, conservation biologists need to be aware of the consequences of applying such models. Although, in conservation biology, land use-focused models are preferable to statistically-focused models, one should always check whether the applied model makes sense on the ground. Predictive models can target mapping efforts towards potentially species-rich sites and permits the incorporation of un-surveyed sites into nature conservancy policies. Species richness distribution maps produced by predictive modelling should therefore be used as pro-active conservation tools.

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Introduction

Species distribution databases are the primary source material used in nature conservation [Dennis *et al.* 1999; Lobo *et al.* 1997]. Ideally, such databases consist of an equal number of visits applied, in a standardized manner, to all the mapping units within the geographical frame where species are recorded. However, most databases are adversely affected by unequal sampling effort in both time and space [Dennis & Thomas 2000] and by differences in the ability of recorders to detect and identify species accurately [Dennis & Hardy 1999], even those of taxonomic groups generally considered as well studied [e.g. birds, mammals and vascular plants - Williams & Gaston 1994]. Nature conservancy policies are mostly based on these incomplete and biased distribution databases [Lobo *et al.* 1997; Samways 1993]. This may lead to non-optimal use of limited resources in nature conservation by wrongly prioritising the designation or acquisition of areas for conservation [Myers *et al.* 2000; Pullin & Knight 2001].
The recently accelerated decline in butterfly diversity in north-west Europe [Dennis & Shreeve 2003; Maes & Van Dyck 2001; Warren *et al.* 2001] calls for a rapid, accurate and cost-effective assessment of species richness over large regions. Recently, several authors have used predictive modelling as a conservation tool, both in poorly-investigated taxonomic groups [e.g. dung beetles - Lobo & Martín-Piera 2002] as in more ‘popular’ groups in countries or regions where large areas have been under-surveyed [Dennis *et al.* 2000, 2002; Fleishman *et al.* 2001a; Sparks *et al.* 1995]. Predictive modelling permits targeting of recorders towards potentially or predicted species-rich areas [Dennis & Hardy 1999], can delimit priority sites for conservation [so called ‘hotspots’, i.e. sites with a large number of species - Myers *et al.* 2000] and facilitates decision making on the impact of land-use changes in un-recorded sites [Fleishman *et al.* 2001a]. This greatly extends the value of collected records in distribution databases and increases the efficiency of mapping schemes that usually have limited logistical and financial resources.

Within the large group of invertebrates, butterflies are certainly the most intensively recorded organisms worldwide and butterfly mapping schemes exist for most of the north-west European countries [Asher *et al.* 2001; Lafranchis 2000; Settele *et al.* 1999; Stoltze 1996; Tax 1989]. But, even with a relatively large number of volunteers, butterfly distribution databases do not overcome the problem of biases in mapping [Dennis & Thomas 2000]. Furthermore, most distribution atlases do not indicate mapping intensity on species distribution maps that enable the reader to interpret distributions [Dennis & Hardy 1999]. High quality data on butterfly distribution and on biotopes, topography and climate are readily available in Belgium [Goffart & De Bast 2000; Maes & Van Dyck 2001]. Some Belgian regions, however, remain poorly surveyed and others are almost certainly under-recorded. Furthermore, recorders generally prefer to visit sites that are known to have a high species richness than to survey new sites where species richness is unknown [Dennis & Thomas 2000].
In this paper [1] we determine land use, topographic and climate factors that correlate with butterfly diversity in Belgium, [2] we develop three predictive models: a statistically-focused, a land use-focused and a hybrid model to predict butterfly diversity using linear modelling and [3] we predict butterfly diversity hotspots using these three models. We compare the results of the three models and discuss their applicability for nature conservancy policy. Although the models are geographically limited to Belgium, we believe that this - from a biodiversity point of view - strongly impoverished north-west European country can be taken as a model area that has some representative character for many other industrialized regions elsewhere in Europe and in the rest of the world.

Materials and methods

Study area

Belgium is a strongly industrialised north-west European country with a high human population density [335 inhabitants/km^2; Van Goethem 2001] and, consequently, intense pressure on nature [OECD 1998]. The general landscape and topography differ considerably between the two administrative regions of Belgium: Flanders and Wallonia. Flanders, the northern part, is a lowland zone [average elevation=38m] and only has a limited total area of nature reserves [1.6% of the territory; Van Goethem 2001]; the most butterfly-rich Flemish biotopes are heathlands and woodlands in the Campine region. Wallonia, the southern part and comparatively an upland region [average elevation=310m] has a similar total area of nature reserves [ca. 1% of the territory; Van Goethem 2001]; here, the most species-rich butterfly biotopes are found on nutrient-poor grasslands and in large woodlands in the Fagne-Famenne-Calestienne and in the Lorraine regions.
The study extends to the whole territory of Belgium [Fig. 4.1]. We used the Universal Transverse Mercator [UTM] projection as this mapping grid is used in all invertebrate recording schemes in Belgium. Units of distribution for the present analyses have a grid size of 25 km^2 [5x5 km], hereafter called 'squares' [N=1374]. For the rest of the analyses, we only consider those squares that have an area of at least 24 km^2 and have >90% of their area within Belgium [N=1108]. The squares of the correction zone of the UTM projection are therefore excluded from the present analyses [see Fig. 4.1].

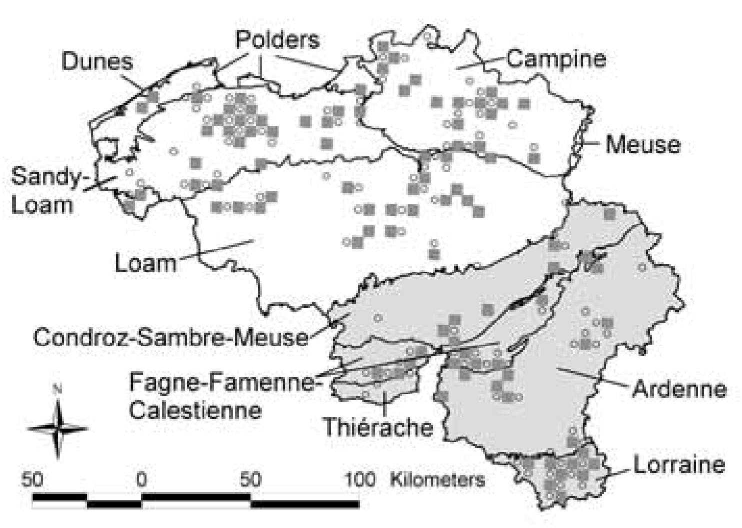

Figure 4.1. Squares used to build three predictive models for butterfly diversity in Belgium: grey squares are used as training set, empty circles as an evaluation set; the eleven ecological regions in Belgium are illustrated. The Atlantic and the Continental biogeographical regions are shown in white and grey respectively.

Butterfly data

Butterfly distribution data were obtained from two separate databases managed by the two regional butterfly working groups: 1] the Flemish database consists of about 210 000 records covering 95% of all Flemish squares [*n* = 644; Maes & Van Dyck 2001] and 2] the Walloon database contains about 50 000 records covering 63% of all Walloon squares [*n* = 802; Goffart & De Bast 2000]. The organization of Flemish and Walloon mapping schemes is separated for several reasons: nature conservancy policy was regionalised in 1980 which means that since then both the Flemish and Walloon government can decide autonomously on nature conservation matters; additionally, species composition differs considerably between the two regions: Flanders has sixty-four indigenous species of which forty-six are still present

[Maes & Van Dyck 2001], Wallonia has 103 native species of which eighty-seven have been recorded since 1990 [Goffart & De Bast 2000]. We excluded all migrant species [*Vanessa atalanta*, *V. cardui*, *Colias croceus*, *C. hyale*] and, where possible, also observations of vagrants [observations not indicative of a breeding population] for the present analyses.

Environmental variables

Three types of 'environmental' data were collected: [1] land use data were derived from the Belgian Corine Land Cover vector map [CEC 1994], [2] topographic variables were derived from a digital elevation model for Belgium [resolution 20m, National Geographical Institute] and [3] climate point data were made available by the Royal Meteorological Institute of Belgium for the period 1996-2001 [Table 4.1]. The areas of land use and topographic variables were estimated per square using ArcView 3.2 [ESRI, Redlands, CA, USA]. In addition, a biotope diversity index [BDI] was estimated per square using only terrestrial biotopes [Shannon diversity index - Magurran 1988]. The length of the edges between grasslands and heathlands on the one hand and deciduous and mixed woodlands on the other was also estimated per square using ArcView 3.2 and Corine Land Cover maps. Since Belgium is located in two European biogeographical regions [EEA 2002b], a binary variable 'region' was incorporated into the analyses: the Atlantic biogeographical region, north of the rivers Meuse and Sambre, and the Continental biogeographical region, south of these rivers [Dufrene & Legendre 1991; Fig. 4.1]. Climate data were interpolated to the squares by universal kriging [Isaaks & Srivastava 1989] when a clear spatial structure could be modelled or, alternatively, using the 'inverse weighted distance' interpolation method. Universal kriging with a linear drift was used to interpolate the point data of yearly cumulated rainfall [mm; n = 186; mean minimum distance between data points 8 km], yearly average maximum temperature [° C; *n* = 114] and yearly cumulated number of frost days [*n* = 114; mean minimum distance between data points 11 km]; 'inverse weighted distance' was used to interpolate the point data of yearly cumulated sunshine exposure [hours, *n* = 22, mean minimum distance between data points 23.5 km].

Table 4.1. Environmental variables, their abbreviation [Abbr.], applied transformation prioir to analyses [Transf.: no = no transformation ; Sqrt = square root transformation; Log = Log_{10}-transformation] and the source of the data.

Name	Abbr.	Transf.	Source
Land use data			
Urban area	Ur	Sqrt	CORINE codes beginning with 1
Arable land	Ar	no	CORINE codes beginning with 2
Deciduous woodland	Dw	Log	CORINE code 311
Mixed woodland	Mw	Log	CORINE code 313
Coniferous woodland	Cw	Log	CORINE code 312
Natural grassland	Gr	Log	CORINE code 321
Heathland and bog	Hb	Log	CORINE code 322+412
Shrub	Sh	Log	CORINE code 324
Salt marshes	Sm	Log	CORINE code 421+423
Dunes	Du	Log	CORINE code 331
Water	Wa	Log	CORINE code 511+512+522
Marsh	Ma	Log	CORINE code 411
Edge	Ed	Sqrt	length of the transistion zone between Gr and Hb on the one hand and Dw and Mw on the other
Biotope Diversity Index	BDI	no	Shannon index of terrestrial biotopes

Climate data	Abbr.	Transf.	Source
Frost	F	Log	Interpolated point data of the RMIB[a]
Rain	R	Log	Interpolated point data of the RMIB[a]
Sun	S	Log	Interpolated point data of the RMIB[a]
Temperature	T	Sqrt	Interpolated point data of the RMIB[a]
Topographic data			
Maximum elevation	El	Sqrt	Digital elevation model Belgium
X co-ordinate	X	no	Lambert Belgium 1972 projection [increases from east to west]
Y co-ordinate	Y	no	Lambert Belgium 1972 projection [decreases from north to south]

[a] Royal Meteorological Institute of Belgium

Analyses

The different predictive models are based on the best-surveyed squares in Belgium during the period 1991-2002. Since mapping intensity differed considerably between Wallonia and Flanders [mean number of visits per square are 10.6 and 66.7 respectively], we applied different selection criteria to determine sufficient recording effort in both regions. To build the predictive models, we selected the best-surveyed squares [75 squares in Wallonia and 122 squares in Flanders] based on the number of visits. Since most of the predictor variables are not normally distributed, we transformed the vari-

ables using either log_{10} or square root functions [resulting in the lowest skewness value] prior to analyses [Table 4.1].
The analysis involved four steps: [1] exploratory analysis of the spatial structure of butterfly diversity, [2] exploratory analysis of the relationships between the predictors in relation to butterfly diversity, [3] the design of three linear models relating butterfly diversity to environmental covariates and [4] application of the linear models to predict butterfly diversity patterns and hotspots in Belgium.
Correlations and linear models statistics are affected by spatial autocorrelation in the response [butterfly diversity] and environmental variables, i.e. the tendency for the value of neighbouring points to be more similar than distant points. In general terms, spatial autocorrelation is important in spatial data analysis for the insight it provides in the data under study [Rossi *et al.* 1992]. It contradicts the assumption of independence among samples replicated through space [Clifford *et al.* 1989; Lennon 2000]. Because of this, modelling the structure of the spatial autocorrelation allows spatial interpolation by the method known as *kriging* [Isaaks & Srivastava 1989].
We have used the experimental co-variogram to quantify spatial autocorrelation in butterfly diversity, which is a function that estimates the level of covariance for points separated by increasingly greater intervals of distance [Rossi *et al.* 1992]. Typically, it is a rising curve [points close by have fairly similar values and a low covariance estimate] that levels off at a given distance known as the 'range' [distance over which sample points are independent], while the height is known as the 'sill'. Points separated by a null distance have a covariance equal to zero, so the curve should start at the origin of the two axes. This is rarely the case with ecological data, and the value at which the experimental semi-variogram intercepts the Y-axis is termed the 'nugget' and represents experimental error or variability at a smaller scale than the smallest distance interval. In the presence of a large-

scale trend, the co-variogram is biased and tends to increase above the limit value of 1. In such a case, the large-scale trend is modelled by a linear or quadratic function of spatial coordinates, and the co-variogram is estimated on the basis of the large-scale trend model residuals. This was the case with butterfly diversity and the co-variogram was estimated using the residuals of a large-scale linear trend model. The co-variogram was modelled by a spherical model using a combination of 'fit-by-eye' and least squares approach, and values for the scale, nugget, range and R^2 of the fit were obtained. Spatial statistics were carried out using the software Surfer 8.0 [Golden software Inc, Golden, USA].

The second step was to explore the relationship between environmental variables and butterfly diversity using two approaches. First, the correlation between butterfly diversity and the land use, topographic and climatic factors was estimated. Unbiased correlation levels of significance were obtained using the method proposed by Clifford *et al.* [1989] modified by Dutilleul [1993] that quantify the reduction in degrees of freedom according to spatial autocorrelation observed in the two variables. Secondly, a principal components analysis was carried out using the set of environmental variables [Table 4.1]; four variables were entered as supplementary to the analysis [butterfly diversity and the number of visits [both $\log_{10}$ values], X and Y co-ordinates distinguishing eastings and northings of grid squares].

The third step was to build linear models relating butterfly diversity to environmental predictors such as frequently applied in similar research [Lobo & Martín-Piera 2002; Luoto *et al.* 2002; McCullagh & Nelder 1989; Nicholls 1989]. Linear models generally assume a constant variance among observations [$Var[e_i] = \sigma^2$], and a covariance among observations equal to zero [$Cov[e_i, e_j] = 0$], which is clearly violated in the presence of spatial autocorrelation. For each linear model, the co-variogram of the residuals was estimated to check if it exhibited evidence of spatial autocorrelation. In such a case the covariance among residuals due to spatial autocorrelation was modelled using the SAS MIXED procedure by the function [Littell *et al.* 1996]

$$Cov(e_i, e_j) = \sigma^2 f(d_{ij})$$

where e_i is the error corresponding to the *i*-th observation, d_{ij} is the distance between the spatial location of the *i*-th and *j*-th residual and *f* is the spatial covariance function. The spatial covariance function was estimated by modelling the experimental co-variogram of the multiple regression residuals using the spherical model [Isaaks & Srivastava 1989]. The spatial covariance model parameters were identified using a combination of 'fit-by-eye' and least squares approaches, selecting the model providing the best fit. The presence of curvilinear relationships between each environmental variable and butterfly diversity was assessed by incorporating the quadratic terms of the environmental variables [Nicholls 1989] and the best function [linear or quadratic] was retained using *Akaike's Information Criterion*; this criterion compromises between model fit [the ability to explain the observed variation on the dependent variable] and model complexity [the number of parameters to estimate; Akaike 1978]. The [linear or quadratic] function of the environmental variable that accounted for the largest reduction in deviance [F-ratio test with *P* level of 0.05] was first incorporated into the model [Crawley 1993; Nicholls 1989]. Next, all the remaining environmental variables were tested in the same way until inclusion was no longer significant. At each step, all previously entered variables were tested for their significance and removed from the model if they were no longer significant. Three multiple regression models were built by splitting the set of 197 squares into two, randomly selected, subsets [cf. Luoto *et al.* 2002; Pearson & Carroll 1999]: a *training set* of 98 squares and an *evaluation set* of 99 squares [Fig. 4.1] that was used to test the models performances. In the first model, termed *statistically-focused model*, environmental variables were entered in descending order of statistical significance. The second model, termed *land use-focused model*, was based on *a priori* knowledge of the relationship between butterfly species richness and the following land use variables [Mac Nally 2000, 2002]:

- biogeographical region: butterfly diversity is markedly higher in the continental region than in the Atlantic region [Goffart & De Bast 2000];
- Biotope diversity index: butterfly diversity increases with increasing biotope diversity [e.g. Hawkins & Porter 2003; Kerr 2001; Kerr *et al.* 2001; Sparks *et al.* 1995; Weibull *et al.* 2000] and the length of the edges between mixed and deciduous woodlands on the one hand and grasslands, heathlands and bogs on the other;
- deciduous and mixed woodland, natural grasslands, heathlands and bogs: all these biotope types are inhabited by typical species giving rise to a higher butterfly diversity [van Swaay & Warren 1999];
- urban and agricultural area: both types of land use have a negative impact on butterfly diversity [Blair & Launer 1997; Dennis & Hardy 2001; Hardy & Dennis 1999].

The choice of this modelling approach stemmed from the assumption that factors chosen according to established relationships that include causal pathways produce more robust models. In the third model, termed *hybrid model*, the same variables as in the land use-focused model were forced in the model in a first step, and additional environmental variables were subsequently added in a stepwise selection procedure such as described in the statistically-focused model [cf. Luoto *et al.* 2002].

The last step was to apply these three models to the whole Belgian territory in order to predict the spatial distribution of butterfly diversity and to determine butterfly diversity hotspots in Belgium [i.e. the 5% most diverse squares; Prendergast *et al.* 1993a]. The spatial distribution of the predicted hotspots was compared with the observed hotspots and field knowledge to determine the most adequate model for nature conservancy policies in Belgium.

Results

Factors determining butterfly diversity

Relationships among variables determining butterfly diversity in Belgium are analysed using PCA [Fig. 4.2]. Five components have eigenvalues greater than 1, each accounting for more than 5% of the variance. Cumulatively these five axes account for more than 72% of the variance in the predictor variable set. The first two axes account for 39% and 15% of the variance respectively and are used to illustrate relationships between the variables. Altogether twelve and four variables respectively load modestly [$\alpha > 0.50$] on axes 1 and 2 [Fig. 4.2]. Variables are polarised on both axes. Axis 1 distinguishes eight variables with positive sign [$\alpha > 0.50$, deciduous, mixed and coniferous woodland, shrub, biotope diversity index, number of frost days, rainfall and elevation] from four with negative signs [$\alpha > 0.50$, biogeographical region, urban area, sunshine and temperature]. Axis 2 distinguishes the area of arable land [positive] from most of the other land use variables [heathlands and bogs, shrubs, biotope diversity index, edges, water, marshes, all negative]. Only four variables have their highest and meaningful loadings on additional axes [natural grasslands, salt marshes, dunes and edges]. Communalities on the first five axes indicate the existence of a substantial amount of unique variance in variables; only one variable [elevation] has communalities higher than 90% on the first five axes and only five variables have substantial variance accounted for [>70%] on the first two axes [biogeographical region, number of frost days, rainfall, temperature and elevation]. Butterfly diversity and the number of visits have 53% and 38% of their variances accounted for on the first five axes. Thus, on this variable set, component scores do not provide an adequate substitute for the original variables in determining variance in butterfly diversity. The plot of variables in the first two axes [Fig. 4.2] indicates a strong geographical patterning to the environmental variables, with X and Y co-ordinates polarised on axis 1. Because of increasing altitude in southern and eastern Belgium the usual environmental trends in northern latitudes is reversed [Dennis 1993; Dennis & Williams 1986; Kerr 2001; Kerr *et al.* 1998]. Conditions become warmer and sunnier to the north and colder and wetter to the southeast. Many natural biotopes also increase to the southeast, especially woodland biotopes. Butterfly diversity and the number of visits are also polarised. Butterfly diversity increases to the southeast on higher ground and decreases to the north, whereas visits decrease eastwards and increase northwards.

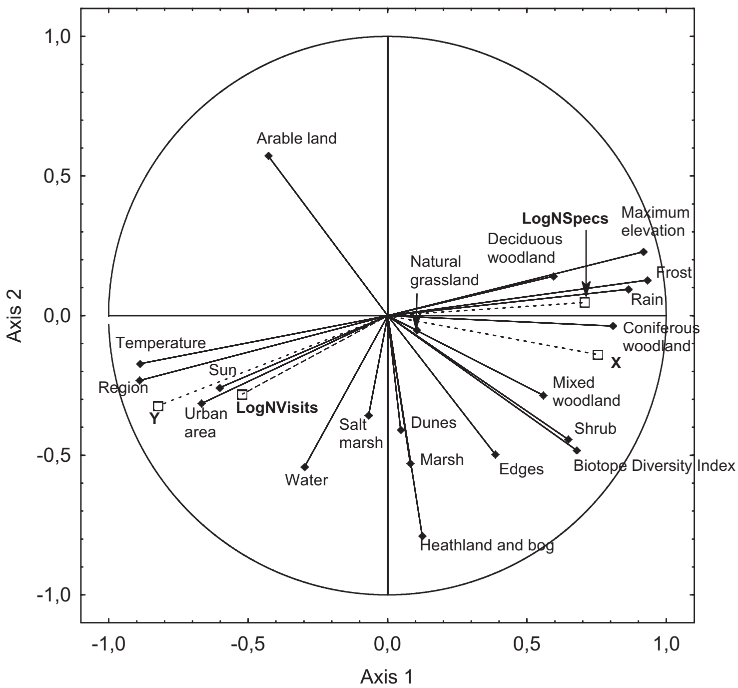

Figure 4.2. Principal components analysis of the environmental variables in the well-surveyed squares [N=197] in Belgium. Axis 1 and 2 explain 39% and 15% of the variation in the data respectively. The number of visits [LogNVisits], the number of species [LogNSpecs] [both log10-transformed] and the X and Y co-ordinates are entered as supplementary variables.

A strong spatial autocorrelation in butterfly diversity was identified at the scale of this study [Fig. 4.3a]. A range distance of 96 km was observed which means that only observations separated by distances >96 km can be considered statistically independent. This high level of autocorrelation had a strong impact on the level of significance of the correlations between butterfly diversity and environmental factors [Table 4.2]. This test should be interpreted cautiously, as it does not necessarily imply that these correlations are spurious, but only that they couldn't be statistically proven within the scale of Belgium. Similarly, butterfly diversity in the well-surveyed squares was not significantly correlated with environmental variables when corrected for spatial autocorrelation; only deciduous and coniferous woodland showed a positive trend with butterfly diversity. Mutually significant correlations are mainly found between land use variables, but not between climatic and topography variables. Arable land is negatively correlated with most other biotope types, while the biotope diversity index is positively correlated with most land use types, except for arable land [Table 4.2].

Table 4.2. Pearson r correlation [lower left part of the table] between the 21 variables and P-value [upper right part of the table] for the well-surveyed grid cells in Belgium [n = 197]. P-values are based on the modified t-test of Dutilleul [1993]. Significant correlations [P < 0.05] are in bold. Freq. = number of squares in which the biotope type occurs. Abbreviations of the environmental variables are given in Table 4.1.

	Freq.	Ur	Ar	Dw	Mw	Cw	Gr	Hb	Sh	Sm	Du	Wa	Ma	Ed	BDI	F	R	S	T	El	X	Y	SR
Ur	197	-	0.258	0.109	0.107	0.189	0.288	0.409	0.143	0.304	0.672	0.052	0.720	0.247	0.286	0.136	0.124	0.274	0.129	0.172	0.449	0.146	0.129
Ar	197	-0.175	-	0.057	**0.005**	**0.008**	0.088	**<0.001**	**0.001**	0.116	**0.013**	0.083	0.051	**<0.001**	**0.002**	0.222	0.157	0.359	0.186	0.267	0.101	0.454	0.137
Dw	141	-0.470	-0.312	-	0.570	0.104	0.279	0.160	0.309	0.210	0.904	0.449	0.765	0.779	0.064	0.207	0.207	0.445	0.298	0.213	0.364	0.132	0.067
Mw	112	-0.345	**-0.377**	0.116	-	**0.029**	0.934	0.158	**0.005**	0.351	**0.010**	0.278	0.154	0.061	**<0.001**	0.111	0.080	0.464	0.102	0.283	0.114	0.588	0.361
Cw	142	-0.453	**-0.464**	0.507	**0.481**	-	0.299	0.818	0.080	0.174	0.752	0.102	0.487	0.057	**0.010**	0.126	0.181	0.168	0.158	0.172	0.204	0.257	0.088
Gr	4	0.098	-0.134	0.103	-0.007	0.096	-	0.684	0.300	0.888	0.829	0.837	0.507	0.189	0.110	0.922	0.874	0.509	0.574	0.245	0.402	0.444	0.212
Hb	44	0.100	**-0.400**	-0.163	0.149	0.028	-0.030	-	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.003**	**<0.001**	**0.003**	0.798	0.664	0.923	0.769	0.923	0.196	0.457	0.986
Sh	65	-0.347	**-0.480**	0.243	**0.496**	0.461	0.087	**0.350**	-	0.087	**0.005**	0.910	**0.004**	**0.005**	**0.001**	0.169	0.096	0.269	0.085	0.180	0.123	0.350	0.272
Sm	1	0.072	-0.112	-0.089	-0.066	-0.093	-0.010	**0.269**	0.120	-	0.912	**<0.001**	**0.003**	0.770	0.621	0.358	0.580	0.156	0.377	0.160	0.693	0.222	0.600
Du	3	-0.038	**-0.178**	0.010	**0.198**	0.028	-0.016	**0.286**	**0.215**	-0.008	-	0.466	0.605	**0.011**	**0.008**	0.987	0.716	0.204	0.324	0.509	0.489	0.241	0.921
Wa	57	0.344	-0.170	-0.122	-0.132	-0.285	0.017	**0.383**	0.015	**0.282**	0.056	-	**0.007**	0.932	0.925	0.223	0.162	0.136	0.089	0.145	0.821	0.143	0.369
Ma	25	0.038	-0.166	0.030	0.146	0.073	-0.049	**0.252**	**0.268**	**0.208**	0.038	**0.227**	-	**0.037**	**<0.001**	0.773	0.713	0.097	0.703	0.770	0.284	0.778	0.560
Ed	21	-0.016	**-0.357**	0.039	0.211	0.287	0.099	**0.677**	**0.346**	-0.021	**0.183**	-0.008	**0.163**	-	**0.009**	0.199	0.109	0.238	0.067	0.159	0.106	0.385	0.202
BDI	197	-0.280	**-0.479**	0.457	**0.617**	**0.673**	0.136	**0.328**	**0.643**	0.035	**0.214**	0.013	0.418	**0.340**	-	0.147	0.163	0.439	0.204	0.259	0.123	0.385	0.166
F	197	-0.664	-0.289	0.545	0.475	0.729	0.010	0.038	0.493	-0.060	0.002	-0.267	0.036	0.255	0.551	-	0.124	0.184	0.136	0.183	0.269	0.224	0.149
R	197	-0.598	-0.296	0.477	0.465	0.572	0.016	0.060	0.508	-0.038	-0.034	-0.277	0.045	0.277	0.468	0.842	-	0.258	0.057	0.182	0.337	0.165	0.225
S	197	0.331	0.155	-0.220	-0.164	-0.435	-0.057	0.012	-0.269	0.098	0.105	0.222	0.178	-0.168	-0.205	-0.575	-0.435	-	0.090	0.092	0.235	0.215	0.338
T	197	0.614	0.269	-0.398	-0.430	-0.612	-0.056	-0.039	-0.520	0.060	0.093	0.345	0.042	-0.312	-0.426	-0.834	-0.845	0.608	-	0.100	0.323	0.182	0.274
El	197	-0.638	-0.259	0.548	0.328	0.685	0.120	-0.014	0.481	-0.093	-0.066	-0.330	-0.034	0.276	0.444	0.901	0.783	-0.685	-0.885	-	0.337	0.170	0.231
X	197	-0.344	-0.395	0.398	0.476	0.614	0.078	0.194	0.543	-0.026	0.063	-0.046	0.137	0.319	0.580	0.795	0.598	-0.520	-0.604	0.722	-	0.473	0.359
Y	197	0.652	0.176	-0.636	-0.167	-0.580	-0.082	0.109	-0.342	0.082	0.113	0.326	0.034	-0.171	-0.343	-0.852	-0.792	0.533	0.780	-0.908	-0.568	-	0.167
SR	197	-0.540	-0.264	0.587	0.220	0.616	0.125	0.002	0.299	-0.037	0.009	-0.171	0.069	0.196	0.404	0.716	0.550	-0.323	-0.506	0.636	0.477	-0.696	-

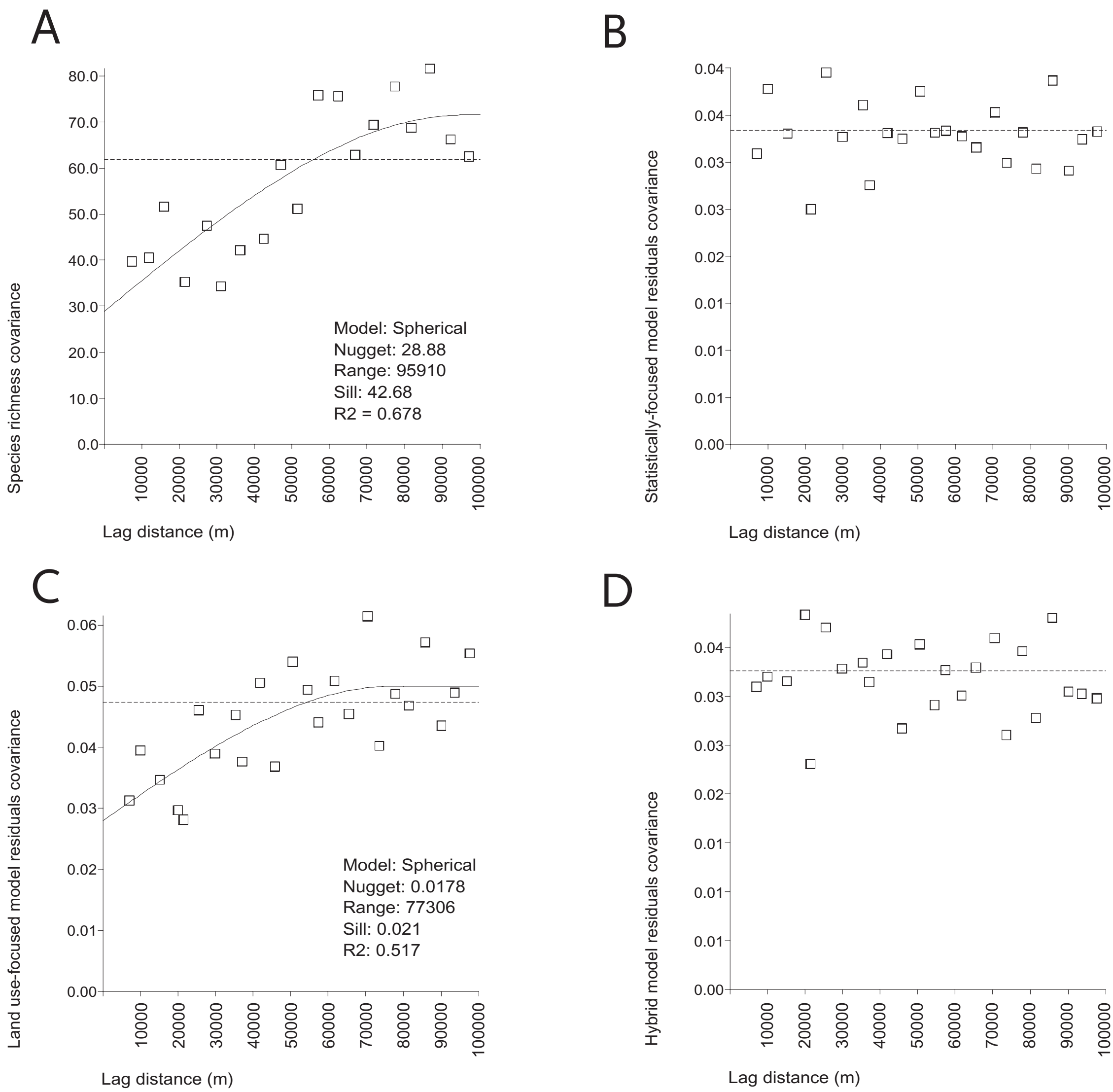

Figure 4.3. Co-variograms of [a] butterfly diversity showing a high degree of spatial autocorrelation up to 95.9 km; [b] the statistically-focused model residuals showing no spatial autocorrelation; [c] the land use-focused model residuals showing spatial autocorrelation up to 77.3 km; and [d] the hybrid model residuals showing no spatial autocorrelation.

Modelling butterfly species richness

When entered separately, 'biogeographical region' accounted for the greatest change in partial R^2 and was entered first into the statistically-focused model. Variable selection ended after the inclusion of deciduous woodland in step 6. Estimates and standard errors of the parameters for the statistically-focused model are given in Table 4.3a. The statistically-focused model explained 77.2% of the variability in the training set [Table 4.3a] and 66.3% of the variability in the evaluation set [Spearman r correlation between observed and expected butterfly diversity in evaluation set= 0.80; $p<0.001$]. The residuals of the statistically-focused model showed no evidence of spatial autocorrelation [Fig. 4.3b] indicating that most of the spatial structure in butterfly diversity [Fig. 4.3a] was accounted for by the environmental factors. It also means that the statistically-focused model did not need to be modified to account for spatial autocorrelation in the residuals.

The land use-focused model explained 67.6% of the variability in the training set [Table 4.3b] and 56.7% of the variability in the evaluation set [Spearman r correlation between observed and expected butterfly diversity in evaluation set= 0.66; $p<0.001$]. Residuals of this model showed a high degree of spatial autocorrelation [Fig. 4.3c], and the linear model therefore included a model of covariance among residuals to obtain unbiased factor estimates and levels of significance.

Finally, the hybrid model explained 77.7% of the variability in the training set [Table 4.3c] and 67.4% of the variability in the evaluation set [Spearman r correlation between observed and expected butterfly diversity in evaluation set= 0.80; $p<0.001$]. Here, residuals showed no evidence of spatial autocorrelation [Fig. 4.3d] indicating that most of the spatial structure in butterfly diversity was accounted for by the complementary environmental factors [the squared terms of elevation, number of frost days and natural grassland area], and that no adjustment was required to account for spatial autocorrelation. Plotting residuals against observed butterfly diversity revealed that the three models all overestimated the number of species for squares with a low observed butterfly diversity and underestimated diversity in species-rich squares [Pearson r between observed species number and the residuals in the training set for the statistically-focused model = 0.47, p<0.001; for the land use-focused model = 0.56, p<0.001; for the hybrid model = 0.45, p<0.001].

Table 4.3. Estimates obtained by the three multiple regression models for butterfly diversity in Belgium: [a] statistically-focused model [R^2 for the training set=0.772; n=98; p<0.001], [b] land use-focused model [R^2 for the training set=0.676; n=98; p<0.001] and [c] hybrid model [R^2 for the training set=0.777; n=98; p<0.001].

A Statistically-focused model	Estimate	SE	df	F value	p
Region	-0.7044	0.1168	91	36.40	<0.001
Cw	0.1337	0.0422	91	10.05	0.002
El^2	-0.0023	0.0004	91	36.03	<0.001
F^2	6.8930	2.5816	91	7.13	0.009
F	-22.4377	9.1307	91	6.04	0.016
Dw	0.0765	0.0383	91	4.00	0.049
Intercept	21.9321				

B Land use-focused model	Estimate	SE	df	F value	p
Region	-0.4987	0.1430	88	12.16	<0.001
Dw	-0.0062	0.0619	88	0.01	0.920
Mw	-0.0548	0.0583	88	0.88	0.350
Gr	0.0273	0.3975	88	<0.01	0.945
Hb	-0.0526	0.0952	88	0.31	0.582
Ed	0.0002	0.0020	88	0.01	0.916
Ur	-0.0269	0.0147	88	3.34	0.071
Ar	-0.0011	0.0007	88	2.47	0.119
BDI	0.0576	0.0857	88	0.45	0.503
Intercept	3.8925				

C Hybrid model	Estimate	SE	df	F value	p
Region	-0.7794	0.1344	85	33.64	<0.001
Dw	0.0265	0.0540	85	0.24	0.625
Mw	-0.0091	0.0515	85	0.03	0.860
Gr	4.9373	2.2077	85	5.00	0.028
Hb	-0.0891	0.0921	85	0.94	0.336
Ed	0.0024	0.0020	85	1.50	0.224
Ur	-0.0278	0.0132	85	4.42	0.039
Ar	-0.0012	0.0007	85	3.50	0.065
BDI	0.0512	0.0803	85	0.41	0.526
El^2	-0.0021	0.0004	85	407.06	<0.001
F^2	0.4999	0.1625	85	162.23	0.003
Gr^2	-11.0284	5.0436	85	144.77	0.032
Intercept	2.8360				

Butterfly diversity and diversity hotspots

Observed butterfly diversity and hotspots

In South Belgium, the Fagne-Famenne-Calestienne and the Lorraine region have the highest butterfly diversity, while the Campine region is the most species-rich region in North Belgium [although absolute numbers are lower in the north - Fig. 4.4a]. Observed butterfly diversity hotspots [i.e. the top 5% of the 1108 analysed squares, N=57 with ≥ 35 species] were all situated in the continental region of Belgium [Fig. 4.5], particularly in the regions Fagne-Famenne-Calestienne [N=23], Lorraine [N=18], Ardennes [N=10], Condroz [N=5] and Thiérache [N=1].

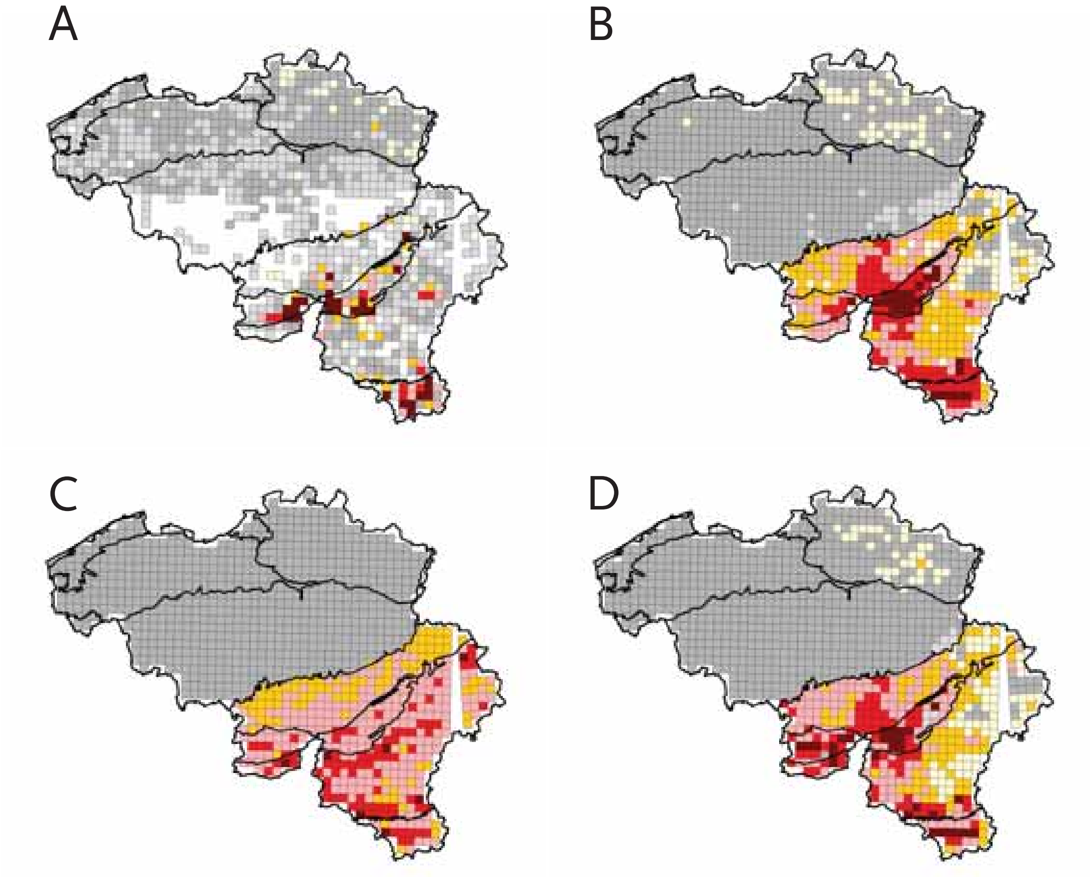

Figure 4.4. Observed [a, unvisited squares are not shown] and predicted butterfly diversity in Belgium using the statistically-focused model [b], the land use-focused model [c] and the hybrid model [d]. Light grey = 1-15 species; dark grey = 15-25 species; yellow = 25-30 species; orange = 30-35 species; rose = 35-40 species; red = 40-45 species and brown = 45 or more species. The blank wedge in the east of Belgium is the correction zone of the UTM projection; the squares in this correction zone are not included in the analyses because their area is much smaller than that of the other squares.

Statistically-focused predicted butterfly diversity and hotspots

The predicted butterfly diversity ranged from 12 to 60 species in the statistically-focused model. Extrapolating the model to the whole of Belgium predicted high butterfly diversity in the Fagne-Famenne-Calestienne, the region around the river Meuse in the Condroz region, in the Lorraine regions and, to a lesser degree, in the Campine region in the north [Fig. 4.4b]. Statistically-focused predicted butterfly diversity hotspots [squares were the model predicted at least 42.4 species] were all situated in the continental region of Belgium [Fig. 4.5], particularly in the regions Fagne-Famenne-Calestienne [N=25], Ardennes [N=13], Lorraine [N=10] and Condroz [N=9].

Land use-focused predicted butterfly diversity and hotspots

The predicted butterfly diversity ranged from 18 to 48 species in the land use-focused model. Extrapolating the model to the whole of Belgium predicted high butterfly diversity in the Ardennes, the Lorraine, the Thiérache region and, to a lesser degree, in the Campine region in the north [Fig. 4.4c]. Land use-focused predicted butterfly diversity hotspots [i.e. squares where the model predicted at least 41.6 species] were all situated in the continental region of Belgium [Fig. 4.5], particularly in the Ardennes [N=39], Fagne-Famenne-Calestienne [N=7], Lorraine [N=7] and Thiérache [N=4].

Hybrid model predicted butterfly diversity and hotspots

The predicted butterfly diversity ranged from 0 to 63 species in the hybrid model. Extrapolating the hybrid model to the whole of Belgium predicted a high butterfly diversity in the Fagne-Famenne-Calestienne, Lorraine, the central part of the Condroz region, in the south of the Ardenne region and, to a lesser degree, in the Campine region in the north [Fig. 4.4d]. Predicted butterfly diversity hotspots using the hybrid model [i.e. squares where the model predicted at least 42.6 species] were all situated in the continental region of Belgium [Fig. 4.5], particularly in Fagne-Famenne-Calestienne [N=25], the Ardennes [N=17], Lorraine [N=8], Condroz [N=4] and Thiérache [N=3].

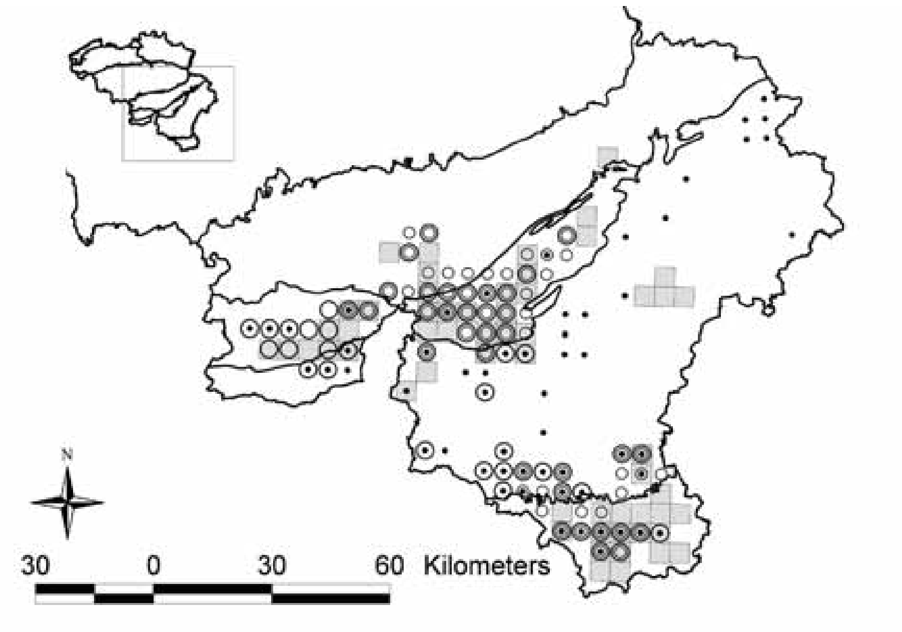

Figure 4.5. Observed [grey squares] and predicted hotspots in the Continental region of Belgium using the hybrid model [large circles], the statistically-focused [intermediate sized circles] and the land use-focused model [small black dots].

Discussion

Factors explaining butterfly diversity

The dominant environmental factors explaining butterfly diversity in Belgium are topography and climate as is often the case in predictive models for butterfly diversity at a large spatial scale [e.g. Fleishman *et al.* 2001a; Kerr *et al.* 1998; Sparks *et al.* 1995]. Although some of the land use variables [different woodland types, shrub, biotope diversity and urban area] have modest to high loadings on the first PCA-axis, they contribute less to the explanation of butterfly diversity in Belgium.

The strong correlation of butterfly diversity with coniferous woodland is unexpected, because this biotope type is not known for its butterfly diversity [van Swaay & Warren 1999] and has an entirely artificial origin in Belgium; an explanation for this correlation can be that conifers were planted in formerly butterfly-rich sites [e.g. heathlands, dry calcareous and wet grasslands, moors; Goffart *et al.* 2000] that have now become too small to be distinguished by CORINE land cover maps.

Modelling butterfly species richness

Mapping the observed butterfly diversity in Belgium [Fig. 4.4a] indicates that not all regions have been surveyed. Fig. 4.4a also shows that North Belgium has been surveyed more completely than South Belgium where a large part of the Loam region and, to a lesser degree, the western part of the Condroz region, are completely un-surveyed. For modelling butterfly species richness in Belgium, we selected the best-surveyed squares in the Flemish and in the Walloon region separately. This implied different thresholds for the number of visits in both regions. The application of a single, common threshold for the whole Belgian territory would have strongly over-represented the Flemish region, which was not appropriate when modelling species diversity for the whole Belgian territory. Furthermore, the predicted number of species per square in both approaches [different thresholds vs. one single threshold] was very strongly correlated reassuring us that the difference in the selection criteria between Flanders and Wallonia did not affect the outcome of the spatial patterns.
The squares to develop and evaluate the model are well spread over the different ecological regions in Belgium [Fig. 4.1]; only three ecoregions are not represented in the training set [Dunes and Meuse are too narrow to have squares that fall completely within Belgium and Thiérache only covers eleven complete squares]. The training set of 98 well-surveyed squares differs significantly from the other squares in a number of variables and is not

a random sample of squares in Belgium [N=1108; MANOVA, F=3.01; p<0.001]. The training set has a smaller area of arable land [F=5.01; p=0.025], but has higher values for salt marshes [F=4.86; p=0.028], marshes [F=13.22; p<0.001] and biotope diversity index [F=7.01; p=0.008]. The training set was selected on the basis of the number of visits and recorders visited these squares more often because they are known to have a high butterfly diversity. This is related to limited areas of arable land and a high biotope diversity [Dennis & Thomas 2000]. Splitting the data set into a training set and an evaluation set is often applied in modelling research, because it provides a more robust estimate of the model appropriateness [e.g., Luoto *et al.* 2002; Pearson & Carroll 1999]. However, when both sets come from the same larger data set, the evaluation set cannot be considered truly independent of the training set [Guisan & Zimmermann 2000], mainly because of spatial autocorrelation [points of the evaluation set are very close to those from the training set]. This was confirmed by the fact that the training and evaluation set did not differ significantly in the environmental variables used or in the number of species or visits [MANOVA: F=0.35; p=0.99]. In a small country like Belgium other types of data set subdivisions would be difficult to achieve. The only way to have independent training and evaluation sets would be to find sampling locations separated by >96 km, which is only possible along a NW-SE axis [choosing the southern sample points from Wallonia on the one hand and the northern sample points from Flanders on the other]. Such a subdivision is not appropriate because it coincides with two different biogeographical regions with different ecological relationships.

All three models explain high percentages of the variability in the evaluation set compared to similar studies [e.g. Lobo & Martín-Piera 2002; Luoto *et al.* 2002]. The hybrid model explains variability in the evaluation set only slightly better than the statistically-focused model but uses more variables resulting in a higher Akaike's Information Criterion value [565.06 vs. 557.84]. In particular, 'biogeographical region' is an important factor explaining butterfly diversity in Belgium [cf. Bio *et al.* 2002]. All three models showed a significant trend when observed diversity was plotted against model residuals. This can be caused by the fact that the model was unable to fully fit the complex interaction between butterfly diversity and environmental variables or by the absence of other predictor variables. Candidates for such missing variables are [1] interaction terms between variables, [2] biotope quality [in the present analyses only biotope quantity is entered] or [3] higher order terms of the environmental variables [Legendre & Legendre 1998]. Inclusion of higher order terms or interaction terms increases the models complexity and makes model interpretations difficult and/or spurious [Bio *et al*. 2002]. Evapotranspiration, one of the frequently used variables in species richness analyses [e.g. Hawkins & Porter 2003], was not incorporated in our analyses, because variation in evapotranspiration is very limited within the extent of Belgium [220 x 270 km - Bultot & Dupriez 1974].

Application of predictive modelling for butterfly conservation in Belgium

Although butterfly distribution is relatively well known in Belgium [Goffart & De Bast 2000; Maes & Van Dyck 2001], predictive modelling can considerably increase the efficiency of butterfly mapping schemes and incorporate un-surveyed regions into nature conservancy policy making. But, the choice of the most accurate predictive model is of major importance when financial and personal resources are limited. Although land use-focused models are usually preferred for their interpretability and logical link with the studied organisms [Lennon 2000; Luoto *et al*. 2002; Mac Nally 2000, 2002], the hybrid and the statistically-focused models predicted butterfly diversity better than the land use-focused one in Belgium. Since regression models aim to minimize residual variance and many of the land use variables covary simultaneously with climatic or topographic variables [Table 4.2], the latter are often better predictors for butterfly diversity than the different land use variables separately [Dennis & Williams 1986]. Both the statistically-focused and the hybrid model explained about 10% more of the variability in the evaluation set than the land use-focused model and are therefore preferred to the latter when used for nature conservation purposes in Belgium.

Hotspots predicted by the statistically-focused, the land use-focused and the hybrid model represent seventy-eight, seventy-five and seventy-seven out of eighty-eight present-day indigenous butterfly species respectively, while observed hotspots cover eighty-two of the present-day indigenous species. Six of the present-day indigenous species are not present in the predicted or observed hotspots: *Heteropterus morpheus, Maculinea alcon, Maculinea arion, Coenonympha glycerion, Coenonympha hero* and *Erebia ligea*. Five of these species are extremely rare and occupy only one [*Heteropterus morpheus, Maculinea arion* and *Coenonympha hero*] or three squares [*Coenonympha glycerion* and *Erebia ligea*] in Belgium, mostly situated along the borders of the country; *Maculinea alcon* is limited to North Belgium [Maes & Van Dyck 1999].

Some authors have predicted species diversity on very large scales using large grid cells [e.g. Kerr 2001; Kerr *et al*. 1998; Lobo & Martín-Piera 2002] Such analyses can provide insight to large-scale differences in species diversity but have practical limitations for nature conservation. Predicting species richness on scales at which species interact with their environment and that are closer to biotope size is therefore more useful in species conservation [Dennis & Hardy 1999; Pearson & Carroll 1999; Prendergast *et al*. 1993b]. This is more likely to be the case using relatively small grid cells [25 km^2 in our case]; further analyses are to be undertaken to determine whether it is possible to extend the present analyses to even smaller grid cells [e.g. 1 km^2].

Predictive modelling is a very useful tool in mapping individual species or species diversity distributions over large and unequally surveyed areas. Land use data, derived from CORINE land cover maps [restricted to Europe] or from satellite images, and other environmental data [e.g. climate, topography] are nowadays readily available for many countries and on different scales. A relatively small set of well-surveyed squares could suffice to apply predictive modelling in under-surveyed regions or countries. For example, applying predictive modelling to the recently published distribution atlas of European butterflies [Kudrna 2002] to indicate potential species distributions, could greatly extend its applicability for European wide nature conservation purposes. We believe that future atlases of butterflies and other organisms, should make more use of predictive modelling to produce predicted distribution maps as a more pro-active conservation tool; predictive models should, of course, always be validated and based on similar well-surveyed regions to produce valuable models that meet minimum standards.

5 SPATIAL COINCIDENCE OF DIVERSITY HOTSPOTS

FOTO: YVES ADAMS

"What many people fail to recognise, and which is therefore a source of endless confusion, is that the establishment of protected areas is not in itself a scientific process. Science may help to inform the process of establishment, but the decisions are ultimately political, ethical, aesthetic, even religious, and embrace much more than just scientific information. At its heart, conservation is not a scientific activity."

John Lawton [1997]. The science and non-science of conservation biology. Oikos 97: 3-5.

The present-day geographic distribution of individual species of five taxonomic groups [vascular plants, dragonflies, butterflies, herpetofauna and breeding birds] is relatively well known on a small scale [5 x 5 km squares] in Flanders [north Belgium]. These data allow identification of areas with a high diversity within each of the species groups. However, differences in mapping intensity and coverage hamper straightforward comparisons of species-rich areas among the taxonomic groups. To overcome this problem, we modelled the species richness of each taxonomic group separately using various environmental characteristics as predictor variables [area of different land use types, biotope diversity, topographic and climatic features]. We applied forward stepwise multiple regression to build the models, using a subset of well-surveyed squares. A separate set of equally well-surveyed squares was used to test the predictions of the models. The coincidence of geographic areas with high predicted species richness was remarkably high among the four faunal groups, but much lower between vascular plants and each of the four faunal groups. Thus, the four investigated faunal groups can be used as relatively good indicator taxa for one another in Flanders, at least for their within-group species diversity. A mean predicted species diversity per mapping square was also estimated by averaging the standardised predicted species richness over the five taxonomic groups, to locate the regions that were predicted as being the most species-rich for all five investigated taxonomic groups together. Finally, the applicability of predictive modelling in nature conservation policy both in Flanders and in other regions is discussed.

Reprinted from Maes D., Bauwens D., De Bruyn L., Anselin A., Vermeersch G., Van Landuyt W., De Knijf G. & Gilbert M. [in press]. Species richness coincidence: conservation strategies based on predictive modelling. *Biodiversity and Conservation*.

Introduction

One of the major challenges for conservation biology is to stop the ongoing and accelerating decline of biodiversity [Pimm *et al.* 1995]. However, limited funding and the constantly growing number of threatened species call for prioritisation. One of the ways to increase efficiency in nature conservation is to direct efforts towards species-rich sites ['biodiversity hotspots' - Myers *et al.* 2000]. This strategy would prevent the extinction of a larger number of species per unit protected area [Reid 1998]. Several authors have delineated the most diverse or most threatened areas world-wide or on a continental scale [e.g., Dobson *et al.* 1997; Pearson & Cassola 1992]. However, most conservation policies are restricted to country or region boundaries and applying the concept of delineating species-rich sites on smaller scales would considerably improve the efficacy of national or local nature conservation policies [Prendergast *et al.* 1993a]. A problem of this approach is that species-rich sites of different taxonomic groups do not necessarily coincide, a finding that calls into question the utility of the concept of 'indicator taxa' for conservation policy purposes [Prendergast *et al.* 1993a; van Jaarsveld *et al.* 1998].

Few countries or regions have sufficiently fine-scaled species distribution data of different taxonomic groups to allow tests for the coincidence of local species richness at a scale where nature conservation is generally applied in the field. Moreover, differences among taxonomic groups in geographic scope of the collected data and in survey efforts can seriously bias a straightforward delineation of species-rich sites [Prendergast *et al.* 1993a, 1999]. Predictive modelling, applying multiple regression techniques on distribution data and a set of environmental variables, has been proposed as a useful tool to 'correct' for differences in mapping intensity and unequal area coverage [Maddock & Du Plessis 1999]. This approach uses data of a limited number of well-surveyed sites to model species diversity for a given taxonomic group as a function of environmental data. After appropriate val-

idation, the model is used to obtain predictions of local species richness, which are less biased owing to differences in mapping intensity and incomplete survey coverage. This method was found to be successful in predicting species richness at different scales for a variety of taxonomic groups: terrestrial vertebrates in American national parks [Edwards *et al.* 1996]; mammals in the North American continent [Badgley & Fox 2000]; butterflies in countries like France [Dennis *et al.* 2002; Dennis & Shreeve 2003], Belgium [Maes *et al.* 2003] or in the Great Basin [Mac Nally *et al.* 2003]. However, these studies were mostly focused on single taxonomic groups and analysing the degree of species richness coincidence among taxonomic groups has so far only been carried out with uncorrected and biased data [Maddock & Du Plessis 1999].
Here, we used fine-scale distribution data [5 x 5 km grid cell size] and the method outlined above to build separate predictive models of five taxonomic groups [vascular plants, dragonflies, butterflies, herpetofauna and breeding birds] in Flanders, accounting for incomplete geographic coverage and variation in survey intensity. Spatial coincidence in the predicted local species richness patterns are explored and discussed in relation to the relevance of biodiversity indicator species and to conservation strategies and policy [e.g., prioritisation of areas for conservation - Myers *et al.* 2000].

Material and methods

Study area

Flanders [total area 13 512 km^2] is one of the federal regions of Belgium, covering the northern part of the country [Fig. 5.1]. It exhibits the typical features of a western industrialised region [OECD 1998]: extensive industry, infrastructure, house building and agriculture, and a very high human population density [431 citizens/km^2 - Van Hecke & Dickens 1994].
Nature conservation is one of the political competences that were trans-

ferred from the Federal to the Flemish Government. The total area of officially recognized nature reserves in Flanders is limited [i.e., 1.6% of the total territory - Decleer & Vanroose 2001]; 1 019 km^2 and 978 km^2 are designated as Habitat Directive [EU Directive 92/43/EEG] and Bird Directive [EU Directive 79/409/EEG] areas respectively, of which 366 km^2 overlap [Dries 2002]. Based on general features of the landscape and geomorphology, twelve ecological regions were distinguished in Flanders [Fig. 5.1 – De Blust 2001].

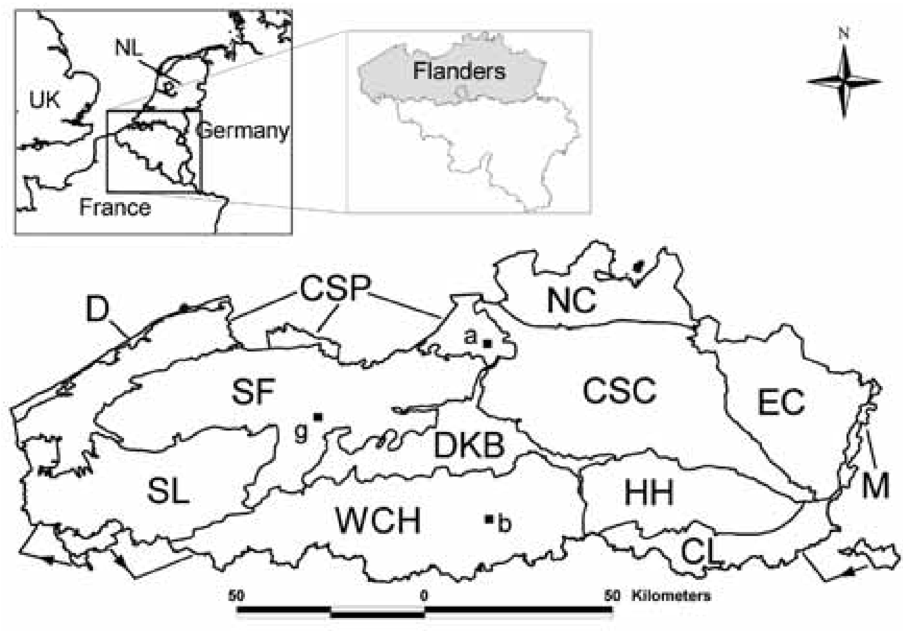

Figure 5.1. Delimitation of the ecological regions in Flanders and geographic location of Flanders within Western Europe and Belgium [insets]. The following ecological regions were considered [De Blust 2001]: Dunes [D]; Sandy Loam [SL]; Sandy Flanders [SF]; Coastal and Scheldt polders [CSP]; Western and Central hills [WCH]; Dender-Klein Brabant [DKB]; North Campine [NC]; Central and South Campine [CSC]; East Campine [EC]; Hageland-Haspengouw [HH]; Calcareous-Loam [CL]; Meuse valley [M]. Also shown is the location of cities of Antwerp [a], Brussels [b] and Ghent [g].

Distribution of species diversity

Data on the distribution of individual species were obtained from different data bases for five taxonomic groups: vascular plants [Van Landuyt *et al.* 2000], dragonflies [De Knijf & Anselin 1996], butterflies [Maes & Van Dyck 2001], herpetofauna [i.e., amphibians and reptiles; Bauwens & Claus 1996] and breeding birds [Vermeersch and Anselin, unpublished data].
Distribution data were collected by a large number of volunteers attributing observations to grid cells of 5x5 km of the Universal Transverse Mercator [UTM] projection for the dragonfly, butterfly and breeding bird mapping schemes and to grid cells of 4x4 km of the "Institut Floristique de la Belgique et Luxembourg" [IFBL] projection for the vascular plants and herpetofauna mapping schemes. Prior to analyses, we converted IFBL grid cells to UTM grid cells, hereafter called squares, by overlaying both projections in the geographical information system Arcview GIS 3.2 [Esri, Redlands, CA]. Only squares having >25% of their area within Flanders were used for analyses [N=585].
Table 5.1 shows some basic information for the different survey schemes. For each taxonomic group, we also obtained information on the number of visits made to each square, allowing assessment of sampling intensity. Distribution records for four taxonomic groups cover at least 90% of the squares subjected to the analyses [Table 5.1a]. The data for dragonflies have a lower coverage [78%], but there is nevertheless a sufficient number of well investigated squares to develop a predictive model for species richness [Lobo & Martín-Piera 2002; Luoto *et al.* 2002].
Species diversity was estimated in each square and for each taxonomic group as the number of indigenous species recorded during the survey period.

Table 5.1. A] Number of records, number of surveyed squares [5 x 5 km; total N = 585], number of indigenous species and the survey period for the different taxonomic groups; B] number of squares in the training [T] and evaluation [E] sets within each of the different ecological regions used to build and evaluate the multiple regression models of five taxonomic groups.

A	Vascular plants	Dragonflies	Butterflies	Herpetofauna	Birds
# Records	2 270 000	40 000	210 000	15 000	323 000
# Investigated squares	585	483	575	563	563
# Indigenous species	1125	58	64	19	163
Survey period	1972-2002	1980-2002	1991-2002	1974-2002	1999-2002

B	T	E	T	E	T	E	T	E	T	E
Calcareous-Loam	7	3	5	2	7	3	7	2	7	3
Central and South Campine	21	8	23	8	23	8	23	9	23	8
Coastal and Schelde polders	11	3	12	6	12	5	11	3	11	5
Dender-Klein Brabant	8	3	8	3	8	3	8	3	8	3
Dunes	1	1	1	1	1	1	1	1	1	1
East Campine	13	3	13	4	12	4	13	4	13	4
Hageland-Haspengouw	10	6	10	4	10	3	10	3	10	3
Meuse	1	1	1	1	1	1	1	1	1	1
North Campine	10	4	10	4	10	3	10	3	9	3
Sandy Flanders	23	9	22	7	23	8	23	9	23	8
Sandy Loam	13	4	14	4	15	5	13	4	15	4
Western and Central hills	26	9	26	10	26	9	26	9	25	9
Total	144	54	145	54	148	53	146	51	146	51

Environmental variables

We extracted data on the distribution of different land uses from the CORINE land cover map for Europe [CEC 1994]. The 44 land use categories distinguished on this map were lumped into 13 land use types that are present in Flanders [Table 5.2]. For each square we estimated the area occupied by the different land use types using the GIS. In addition, we estimated biotope diversity per square using the Shannon diversity index [Magurran 1988]. Climate data were obtained from the Royal Meteorological Institute of Belgium for the period 1996-2001. Point climate data were interpolated in the squares by universal kriging [Isaaks & Srivastava 1989] when a clear spatial structure could be modelled or, alternatively, using the 'inverse weighted distance' interpolation method. Universal kriging with a linear drift was used to interpolate yearly cumulated rainfall [mm; 186 locations], yearly average maximum temperature [°C; 114 locations] and yearly cumulated number of frost days [114 locations]. The 'inverse weighted distance' method was used to interpolate yearly-cumulated sunshine exposure [hours, 22 locations]. Spatial interpolations were carried out using the software Surfer 8.0 [Golden Software Inc., Golden, Colorado, USA]. Topographic variables [mean elevation; elevation range [i.e., the difference between the highest and lowest elevation]] were derived from a digital elevation model for Belgium [1996, National Geographical Institute, resolution 20 m] and estimated for each square using the GIS.

Table 5.2. Symbols and data source for the environmental variables used in the multiple regression models for species diversity.

Variable	Symbol	Data source
Biotope data		
Urban area	Ur	CORINE codes beginning with 1
Agricultural land	Ar	CORINE codes beginning with 2
Deciduous woodland	Dw	CORINE code 311
Mixed woodland	Mw	CORINE code 313
Coniferous woodland	Cw	CORINE code 312
Natural grassland	Gr	CORINE code 321
Heathland and bog	Hb	CORINE code 322+412
Shrub	Sh	CORINE code 324
Salt marshes	Sm	CORINE code 421+423
Dunes	Du	CORINE code 331
Water courses	Wc	CORINE code 511+522
Water bodies	Wb	CORINE code 512
Marsh	Ma	CORINE code 411
Biotope Diversity Index	BDI	Shannon diversity index of biotopes
Climate data		
Number of frost days	F	Interpolated point data of the RMIB[a]
Yearly rainfall	R	Interpolated point data of the RMIB[a]
Sum of sunhours	S	Interpolated point data of the RMIB[a]
Maximum temperature	T	Interpolated point data of the RMIB[a]
Topographic data		
Mean elevation	El	Digital elevation model Belgium
Range elevation	RE	Digital elevation model Belgium

[a] Royal Meteorological Institute of Belgium

Modelling species diversity

Prediction of species diversity over the entire territory of Flanders was based upon multiple regression models developed on a subset of the surveyed squares and relating species diversity to environmental variables. However, multiple regressions require that the predictor variables are mutually independent. Therefore, we first examined the correlations among the twenty environmental variables, using data for all squares [N = 585]. Unbiased correlation levels of significance were obtained using the method proposed by Clifford *et al*. [1989] and modified by Dutilleul [1993] that quantifies the reduction in degrees of freedom due to spatial autocorrelation in the two variables. We also corrected the levels of significance using the multiple testing adjustment procedure of Legendre & Legendre [1998]. In addition, we performed a principal components analysis [PCA] on all environmental variables to examine whether it was appropriate to substitute the original variables by a reduced set of component variables.
We built and evaluated a predictive model per taxonomic group on a subset of the squares [N = ± 200]. Specifically, we selected the 30% best-surveyed [i.e., most visited] squares within each of the twelve ecological regions [Fig. 5.1 and Table 5.1b]. This procedure accounts for differences in area, mapping intensity and species richness in the different ecological regions. Three fourths of these squares were used to build the model [hereafter called *training set*], while the remaining fourth of the squares were used to evaluate the model [hereafter called *evaluation set*]. Attributing the well-surveyed squares to either the training or to the evaluation set was based on a random selection within each ecological region.
Species diversity for each of the five taxonomic groups was modelled using forward stepwise multiple regression. The putative presence of curvilinear relationships between the predictor variables and species richness was

taken into account by incorporating the quadratic terms of the predictor variables [Nicholls 1989]. The function of the independent variable that accounted for the largest reduction in deviance [F-ratio test, $p<0.05$] was first incorporated into the model [Crawley 1993]. Next, all the remaining predictor variables were tested in the same way until inclusion was no longer significant. At each step, all previously entered variables were tested for their significance and removed from the model if they were no longer significant. For each linear model, the co-variogram of the residuals was estimated to check for spatial autocorrelation [Overmars *et al.* 2003]. No spatial autocorrelation was present in the residuals of the predictive models for dragonflies and breeding birds. For vascular plants, butterflies and herpetofauna the covariance among residuals due to spatial autocorrelation was modelled using the SAS MIXED procedure by the function [1] where e_i is the error corresponding to the i-th observation, d_{ij} is the distance between the spatial location of the i-th and j-th residual and f is the spatial covariance function [Littell *et al.* 1996]. The spatial covariance function was adjusted by modelling the experimental co-variogram of the multiple regression residuals using the spherical model [Isaaks & Srivastava 1989]. The spatial covariance model parameters were identified using a combination of 'fit-by-eye' and least squares approaches, selecting the model providing the best fit. The final regression model for each species group was used to predict species diversity within the squares in the evaluation set. We stress that these squares were not used to build the model. We assessed the goodness of fit of the predictive model by the Spearman rank correlation between the predicted and observed species diversity in the squares of the evaluation set. After its validation, the regression model for a species group was used to predict species richness in all squares.

[1]

$$Cov(e_i, e_j) = \sigma^2 f(d_{ij})$$

The mean predicted species diversity in each square was estimated as the mean of the standardised predicted species richness [SSR] over the five tax-

[2] onomic groups; SSR is calculated as [2]

$$SSR = \frac{(x_i - x_{\min})}{(x_{\max} - x_{\min})}$$

where x_i is the predicted species richness in the ith square and x_{min} and x_{max} are the minimum and maximum predicted species richness respectively [Gower 1971].

Results

Correlations of observed species diversity among taxonomic groups

A rather restricted number of squares [N = 244; 38% of total] was surveyed at least three times for all five taxonomic groups together. We used this subset of squares to calculate correlations of the observed species diversity among the five groups [Table 5.3]. The lowest correlation was between plant and butterfly species diversity, while species diversity of dragonflies is highly correlated with that of butterflies and breeding birds.

Geographic patterns of observed species diversity for dragonflies, butterflies, herpetofauna and breeding birds showed a large concentration of species-rich squares in the Campine regions [NE Flanders - Fig. 5.2b-e]. Plant species diversity did not show such a pronounced pattern and species-rich squares were more scattered over Flanders [Fig. 5.2a].

Table 5.3. Pairwise correlation coefficients [Spearman rank correlation] for the observed species diversity among the different taxonomic groups. The correlations are based on data for the 244 squares that were surveyed at least three times for each of the species groups. P values: ** p<0.01; . *** p<0.001.

	Vascular plants	Dragonflies	Butterflies	Herpetofauna
Dragonflies	0.270***	-		
Butterflies	0.174**	0.508***	-	
Herpetofauna	0.286***	0.336***	0.229***	-
Birds	0.306***	0.527***	0.306***	0.286***

Correlations among environmental variables

The analysis of the collinearity among the environmental variables revealed that only 17 out of 179 possible correlations [i.e., 9%] were judged statistically significant after correcting for spatial autocorrelation and multiple testing [Table 5.4]. Biotope diversity was the variable that was most frequently correlated with other environmental characteristics. The area of agricultural land was negatively correlated with the area occupied by most other land use types. It should be noted that very few significant correlations were found between the land use variables and the climatic and topographic variables [Table 5.4].

A principal components analysis on the correlation matrix of all environmental variables [N = 20] yielded six component axes that had eigenvalues > 1; together they represented 65% of the total variance. The extraction of 14 axes was required to retain 90% of the original variance. Hence, the principal components analysis did not achieve a meaningful reduction of the dimensionality of our data set. Therefore, we opted to use the original environmental variables as independent variables in the ensuing multiple regression analyses.

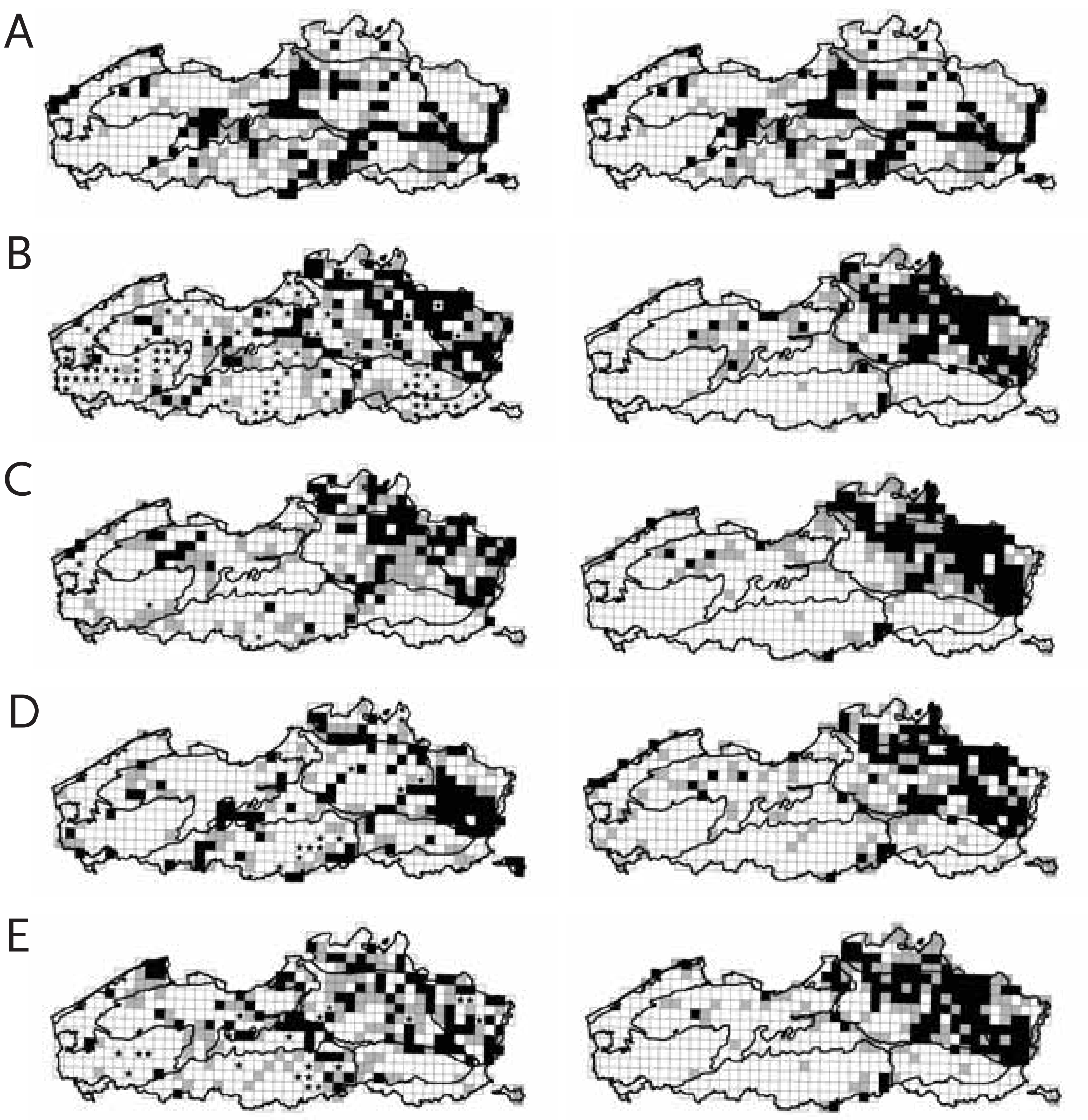

Figure 5.2. Geographic patterns of observed [left] and predicted [right] species richness for the different taxonomic groups: vascular plants [a], dragonflies [b], butterflies [c], herpetofauna [d] and birds [e]. In black the top 100 most species rich squares, in grey the next 100 most species rich squares; stars indicate squares that were not surveyed.

Table 5.4. Pearson correlation coefficients among the twenty environmental variables for all squares in Flanders [N = 585]. In bold: correlations that were significant [P < 0.10] after applying a modified t-test that accounts for spatial autocorrelation and a correction for multiple testing. Freq = number of squares in which the biotope type occurs. . = insufficient data. Abbreviations of the environmental variables are given in Table 5.2.

	Freq	Ur	Ar	Dw	Mw	Cw	Gr	Hb	Sh	Sm	Du	Wc	Wb	Ma	BDI	F	R	S	T	El
Ur	584	-																		
Ar	584	**-0.472**	-																	
Dw	340	0.113	-0.197	-																
Mw	203	**-0.238**	-0.258	-0.066	-															
Cw	262	-0.036	**-0.332**	-0.008	0.196	-														
Gr	8	0.388	-0.563	.	.	.	-													
Hb	139	-0.049	-0.231	-0.070	0.149	0.350	0.145	-												
Sh	101	-0.016	**-0.427**	0.012	0.079	0.321	.	**0.517**	-											
Sm	8	-0.470	0.063	.	.	.	-0.997	0.279	.	-										
Du	22	0.191	**-0.588**	0.013	-0.328	0.225	0.761	0.178	0.003	.	-									
Wc	83	0.046	-0.182	-0.072	0.115	0.230	0.335	0.333	0.132	0.666	-0.832	-								
Wb	175	0.109	-0.186	-0.005	-0.184	-0.020	-0.719	0.155	0.043	-0.655	0.291	-0.050	-							
Ma	60	0.069	-0.197	0.100	0.124	-0.323	.	0.150	0.403	.	-0.834	0.223	0.046	-						
BDI	585	**0.473**	**-0.476**	**0.356**	**0.536**	**0.546**	0.812	**0.447**	**0.625**	0.515	0.378	0.348	0.182	**0.522**	-					
F	585	0.233	-0.259	0.150	**0.330**	0.315	0.283	0.175	0.282	0.327	-0.297	-0.132	0.003	0.208	0.578	-				
R	585	-0.045	-0.123	-0.004	0.188	0.140	0.267	0.122	0.244	0.126	-0.370	0.198	-0.013	0.158	0.225	0.179	-			
S	585	-0.283	-0.036	-0.046	0.025	0.125	-0.120	0.007	0.194	0.057	0.201	0.033	0.195	0.101	-0.095	-0.194	0.020	-		
T	585	0.188	-0.036	-0.037	0.289	-0.006	0.192	0.157	0.204	0.166	-0.287	0.276	-0.011	-0.013	0.321	0.397	0.044	-0.211	-	
El	585	0.130	-0.012	0.094	**0.299**	0.201	-0.251	0.025	0.039	-0.423	-0.389	-0.228	-0.170	0.038	0.133	0.439	-0.338	-0.390	0.162	-
RE	585	0.133	0.024	0.125	0.098	0.083	0.047	-0.062	-0.046	-0.328	-0.182	-0.152	-0.196	-0.031	-0.049	0.096	-0.420	-0.360	-0.003	**0.837**

The predictive models [i.e., multiple regression analyses] for all five species groups were highly significant [R^2 varies between 37 - 66%; Table 5.5]. More important, the Spearman rank correlations between predicted and observed species richness in the squares of the evaluation set were also highly significant [Table 5.5], indicating that the regression models provided reasonably accurate predictions of species diversity in the evaluation set.

Table 5.5. Parameter estimates for the environmental variables [linear and/or quadratic terms] that entered in the multiple regression analyses of species diversity within each species group. The number between brackets denotes the order in which the variable was entered into the model. Codes for the environmental variables are given in Table 5.2. P values: * P < 0.05; ** P < 0.01; *** P < 0.001. n.s. not significant.

	Vascular plants	Dragonflies	Butterflies	Herpetofauna	Birds
Ur	0.061***[4]	-	-	-	-
Ur^2	-	-	-	-0.010*[2]	-
Ar	-	-0.074*[8]	-	-	-
Dw	-	0.022*[9]	-	0.070**[3]	-
Dw^2	0.005***[3]	-	-	-0.018*[4]	-
Mw	-	0.043***[2]	-	-	0.012**[3]
Mw^2	-0.001$^{n.s.}$[2]	-	0.007***[1]	-	-
Gr^2	0.015***[5]	-	-	-	-0.013**[2]
Wc	-	-0.028*[7]	-	-0.019*[5]	-
Wb	0.011***[6]	-	-	-	-
Wb^2	-	0.015***[3]	-	-	0.005**[4]
BDI	0.240***[1]	0.659**[1]	0.320***[2]	-	0.297***[1]
BDI^2	-	-	-	1.269***[1]	-
S	-1.126***[7]	-	-	-	-
S^2	-	-0.573***[4]	-	-	-
T	-2.104$^{n.s.}$[8]	-	-	-	-
RE	-	0.414**[6]	-	-	-
RE^2	-	-0.296**[5]	-	-	-
Model R^2	0.656***	0.596***	0.442***	0.368***	0.475***
Spearman r evaluation set	0.639***	0.779***	0.648***	0.492***	0.454***

The number of variables used to build the predictive models ranged from two [butterflies] to nine [vascular plants]. Biotope diversity was the only variable that entered in the models for all five species groups, either as a linear or as a quadratic term [Table 5.5]. Residuals of the models for vascular plants, butterflies and herpetofauna showed some degree of spatial autocorrelation and the models therefore included a model of covariance among residuals to obtain unbiased estimates and levels of significance [Keitt *et al.* 2002].

We applied the models to obtain estimates of species diversity for each species group over the total area of Flanders [Fig. 5.2]. Species richness of vascular plants was predicted to be high in the dune areas, near the cities of Ghent and Antwerp, in the transition zones between several ecological regions and in the valleys of the rivers Dijle and Scheldt [Fig. 5.2a]. Species diversity of dragonflies, butterflies, herpetofauna and breeding birds was predicted to be high mainly in the Campine regions and in some scattered squares in the Sandy Flanders region [Fig. 5.2b-e].

Correlations of the geographic pattern of predicted species diversity among the five species groups is given in Table 5.6. On average, correlations among the four faunal groups were clearly higher than correlations between each of the faunal groups and predicted plant species richness [average Spearman r among the four faunal groups = 0.789; average Spearman r of vascular plants with the four faunal groups = 0.562].

The distribution pattern of the mean predicted species diversity for all squares in Flanders showed a prominent concentration of species-rich squares in the Campine regions [Fig. 5.3]. Other areas with a less pronounced aggregation of squares with a predicted high mean species diversity were found in the Dunes region, the south-eastern part of the Western and Central Hills region, the northern part of the Dender - Klein Brabant region and scattered over the Sandy Flanders region [Fig. 5.3].

Table 5.6. Pairwise correlation coefficients [Spearman rank correlation] for the predicted species diversity among the different taxonomic groups. The correlations are based on predicted species richness in all squares [N = 585]. Between brackets: the number of squares in common in the top 100 most species-rich squares. P values: *** p<0.001.

	Vascular plants	Dragonflies	Butterflies	Herpetofauna
Dragonflies	0.563*** [35]	-		
Butterflies	0.589*** [24]	0.872*** [77]	-	
Herpetofauna	0.489*** [33]	0.656*** [75]	0.707*** [77]	-
Birds	0.606*** [37]	0.898*** [79]	0.937*** [76]	0.666*** [69]

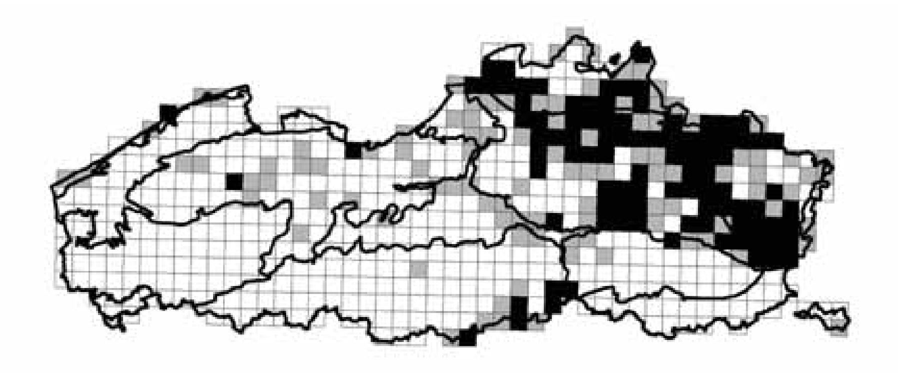

Figure 5.3. Geographic pattern of the mean standardised predicted species richness over the five taxonomic groups. In black the top 100 most species rich squares, in grey the next 100 most species rich squares.

Discussion

Despite considerable efforts that were put into the separate survey schemes, only a relatively low number of squares was adequately surveyed for all of the species groups in common. This hampers exploring the coincidence among species groups in the geographic patterns of observed species diversity. Therefore, we adopted an alternative approach using predictive modelling of species diversity. We here discuss some methodological aspects, the main results of the predictive models and the relevance of our findings for nature conservation.

Reliability of species distribution data

Our approach ideally requires the simultaneous collection of data on species diversity for each of the taxonomic groups and for the environmental variables. Although collecting periods for the different data sets inevitably differed, they overlap largely [Table 5.1], such that it is reasonable to assume that any discrepancies did not affect the outcome of our analyses.
The largest difference in survey periods was between breeding birds, which were studied recently and in a short time-span [i.e., 1999 - 2002], and the other species groups. For the latter taxa, it was necessary to lump information collected over a longer period to obtain adequate geographic coverage of the data. A drawback of accumulating survey data over long periods is that they may include data on species that went extinct after the initial years of the mapping period. Hence, for the taxonomic groups that were surveyed over relatively long time periods [i.e., vascular plants, dragonflies and herpetofauna], our data may have overestimated the present-day species diversity in some of the squares. However, the majority of the recent local species extinctions in Flanders occurred in the period 1950 - 1970 [Bauwens 1999; Bauwens & Claus 1996; De Knijf & Anselin 1996; Maes & Van Dyck

2001], i.e., before the start of the mapping schemes of the taxonomic groups studied here. Hence, the limited number of local species extinctions that took place during the survey periods should not have had a substantial impact on our estimates of local species richness.
Geographic variation in sampling intensity is inevitable in survey schemes carried out by volunteers and may induce biases in the analyses. To minimize such biases, we built and evaluated the predictive models using data from the most frequently visited squares within the different ecological regions of Flanders. Moreover, the numbers of squares included in the analyses were proportional to the area of each of the ecological regions, such that the selected squares were distributed homogenously over Flanders. This procedure reduces any biases induced by geographic variation in local species richness.

Modelling species diversity

Nature conservancy policy makers throughout the world have to base conservation strategies on incomplete and/or biased data [Lobo *et al.* 1997; Samways 1993], even in relatively well-surveyed countries or regions such as NW Europe [Dennis & Hardy 1999]. Bias in the available data is caused by the unequal distribution of recording intensity [Dennis & Thomas 2000; Dennis *et al.* 1999]. This may lead to non-optimal use of limited resources in nature conservation by wrongly prioritising the designation or acquisition of areas for conservation [Myers *et al.* 2000; Pearson & Carroll 1998].
Through modelling techniques, we can upgrade biased and incomplete distribution data bases by assessing the potential conservation value of unsurveyed or clearly under-surveyed sites [Lobo *et al.* 1997; Maddock & Du Plessis 1999].
The multiple regression analyses indicated that geographic variation of

species diversity within each taxonomic group could be explained, albeit to a variable degree, by geographic variation in environmental variables in Flanders. Local species richness of all five taxonomic groups was more often correlated with land use variables than with climatic or topographic variables, a result that contrasts other predictive models of species richness [e.g., Fleishman *et al.* 2001c; Sparks *et al.* 1995]. Compared to large-scale studies [e.g., continents] where variation in climate and topography is far more pronounced [e.g., Badgley & Fox 2000; Kerr *et al.* 1998], Flanders has little geographic variation in climate [e.g., mean maximum temperature ranges from 13.7 - 15.1°C] and topography [elevation ranges from 1 - 237m]. Our analyses revealed that species richness of all five taxonomic groups was positively correlated with biotope diversity. This finding emphasizes the importance of the presence of different biotopes for species richness [Kerr & Packer 1997; Weibull *et al.* 2000].

To be reliable for nature conservation purposes, predictive modelling should always include a testing phase, preferably using an evaluation data set that is independent from the data used to build the models [Mac Nally 2000]. Thus, we compared predicted to observed local species richness for a different set of squares, which were selected using the same criteria as for the selection of the squares in the training set. Complete independence between the training and evaluation set was probably not achieved here as both sets of squares were extracted from the same larger data set [Guisan & Zimmermann 2000]. However, the restricted area of small regions like Flanders impedes achievement of truly geographic independence between training and evaluation data sets.

Correlations between observed and predicted species diversity were highly significant in all five species groups. The correlations were particularly high for the models of vascular plants, dragonflies and butterflies and the geographic patterns of observed and predicted local species richness patterns

largely coincided. Although significant, correlations between observed and predicted species diversity were lower for herpetofauna and birds. Species richness of the herpetofauna may be more difficult to model because of the low number of species involved [N = 19] and the relatively large among-species differences in habitat preferences. The relatively large scale on which birds interact with their environment – vagrancy is higher in birds than in the other taxonomic groups – may make it more difficult to build a predictive model with the environmental variables used.
The significant rank correlations showed that the predictive models produced acceptable estimates of the rank order of species diversity within each species group. However, the distribution of the models residuals indicated that the models systematically underestimated the number of species in species-rich squares and overestimated species diversity in species-poor squares [cf. Lobo & Martín-Piera 2002]. This indicates that the predictive models are not fully able to fit the interactions between local species richness and the environment on the scale used here [grid cells of 5 x 5 km]. This can be due to the high degree of fragmentation of the Flemish landscape [EEA 2002a] which renders predictive modelling more difficult, or to variables not accounted for in the present model [Pape Moller & Jennions 2002]. Possible missing variables are interaction terms between variables, biotope quality [in the present analyses only biotope quantity is entered] or higher order terms of the environmental variables [Legendre & Legendre 1998]. Inclusion of higher order terms or interaction terms increases the models complexity and makes model interpretations difficult and/or spurious [Bio *et al.* 2002]. However, this does not invalidate our approach for conservation-oriented applications, which are based on relative differences in species diversity among areas, rather than on absolute numbers.

Coincidence of species diversity among taxonomic groups

Globally, reliable distribution data are available for at most a limited number of taxonomic groups. The lack of data for 'unpopular' species groups usually results in nature conservation strategies that are based upon data for a limited number of taxa [Prendergast *et al.* 1993a]. To overcome the problem of time-consuming – and hence expensive – surveys for a wide range of taxonomic groups, conservationists and policy makers apply the concept of indicator taxa, where one taxon is used as a surrogate for many others [Caro & O'Doherty 1999; Simberloff 1998]. In many countries and regions, birds and vascular plants have been used as indicator taxa [e.g., Bibby 1999; Blair 1999; Niemela & Baur 1998; Pharo *et al.* 1999]. However, different studies have shown that the coincidence of species richness across taxa can be very low [e.g., Andelman & Fagan 2000; Prendergast *et al*. 1993a; van Jaarsveld *et al*. 1998]. These results call into question the reliability of the concept of indicator taxa for conservation purposes.

Our results indicate that correlations of geographic patterns of both observed and predicted species-richness among species groups were relatively high in Flanders compared to those reported in other studies carried out at larger [e.g., Prendergast *et al*. 1993a; van Jaarsveld *et al*. 1998] or finer scales [e.g., Vessby *et al*. 2002]. This does not appear to be related to the study scale, but more likely to the high pressure on land use which constrains species distribution to a restricted number of semi-natural sites in Flanders. The coincidence of the predicted local species richness was especially high among the four faunal groups, which had on average 76 squares in common among the top 100 most species-rich squares. This indicates that, in Flanders, the four investigated faunal groups can be used as fairly good indicator taxa for one another. On the other hand, the geographic coincidence in predicted species diversity is much lower between vascular plants and each of the four faunal groups, with on average only 32 squares in common among the top 100 most species-rich squares. Hence, plant species diversity cannot be considered as a useful indicator of faunal species richness within Flanders or vice versa.

Prioritising areas for conservation

The prioritisation and subsequent designation of areas for conservation often lacks quantitative scientific underpinning and is frequently based on 'best professional judgements' or on personal experience of local conservationists [Pullin & Knight 2001]. For instance, until present, no attempts were made to integrate overall analyses of species distribution data into the designation of important conservation areas in Flanders. Rather, designation of most conservation areas in Flanders was based on the presence of certain [threatened] biotopes, with strong constraints imposed by political and socio-economic considerations. It should therefore be questioned to what extent these conservation policies are expected to contribute to the objective of preserving species diversity at its present-day level. To answer this question, we explore the extent of geographic overlap between recently designated [complexes of] conservation areas and the patterns of predicted local species richness.

A first conservation policy that was recently [2001] implemented in Flanders was the designation of ca. 1630 km^2 as 'Natura 2000' areas, in compliance to the Habitat and Bird Directives of the European Union. The 'Natura 2000' network aims at preserving species diversity on a European scale and prioritises the conservation of internationally threatened species and biotopes. Because very few internationally threatened species occur in Flanders, the designation of 'Natura 2000' areas was primarily based on the presence of certain biotope types. Overlays of the map of designated

‘Natura 2000’ sites with the map of the mean predicted local species richness revealed that only 43 squares of the top 100 predicted most species-rich squares overlapped with the squares that contained at least 500 ha of ‘Natura 2000’ sites.

The regional government of Flanders recently [2003] also approved the designation of ca. 900 km^2 of conservation areas to create the ‘Flemish Ecological Network’ [FEN]. One of the explicit objectives of this policy is the maintenance of the present-day species richness in Flanders. Nevertheless, selection of the FEN areas was mainly based on the presence of certain biotopes and no systematic consideration was given to species diversity. Overlays of the map of designated FEN sites with the map of the mean predicted local species richness revealed that only 46 of the top 100 predicted most species-diverse squares overlapped with the squares that contained at least 500 ha of FEN sites.

Although both conservation programmes [Natura 2000 and FEN] differ greatly in the geographic scale of their objectives, their designated areas overlap to almost identical extent with sites with the mean predicted species richness in Flanders. This is presumably a consequence of the fact that both programmes used similar criteria to designate areas, even though they have diverging objectives. Second, although the designated areas overlap to some extent with the sites with a high mean predicted local species richness, less than one half of the predicted most species-rich squares was adequately incorporated in the schemes. Consequently, we question whether the ‘Flemish Ecological Network’ will achieve its objective of maintaining local species diversity at its present-day levels.

This example illustrates how decisions on the designation of conservation areas may greatly benefit from predictive modelling performed at a local scale. We strongly believe that policy makers in Flanders, but also in other parts of the world, should make more use of modelling techniques to produce predicted maps of species richness among taxa as a pro-active conservation tool because it allows to better target sites with a collective high species richness for different taxonomic groups. Furthermore, the simultaneous use of taxonomic groups representing organisms at different trophic levels [from nectar feeders to predators] and from both aquatic and terrestrial biotopes at different levels of geographic scale [from very small [vascular plants, invertebrates] to very large [birds of prey], assures a representative sample for a wide variety of other un-investigated taxa [cf. Vanderklift *et al*. 1998]. Different avenues for application in future conservation programmes are under investigation at present: the prediction of local species richness on an even smaller scale than the one presented here [e.g., 1 x 1 km squares or in the optimal scenario, parcels; Fleishman *et al*. 2003b] and the incorporation of taxonomic groups for which distribution data are less complete than for the ones studied here. Detailed land cover classification will become more readily available through remote sensing, such that it will become possible to perform similar analyses on relatively large regions [Kerr & Ostrovsky 2003; Turner *et al*. 2003]. Future analyses should further explore the minimum number of squares and survey visits needed to adequately model species richness in a given region.

6 Ants in wet heathlands, a biotope of European conservation concern

foto: Jeroen Mentens

"… science is clearly involved in delivering effective management once management goals have been defined. … all management practices require an underpinning of ecological science, both to carry them out effectively and to predict their consequences. Science can also inform managers, politicians, or citizens of the consequences of continuing with some particular course of action, or changing or stopping it, and hence can help to set management objectives …"

John Lawton [1997]. The science and non-science of conservation biology. Oikos 97: 3-5.

During a survey of 23 wet heathland sites in Flanders [north Belgium] in 1999 and 2000, using both manual nest searching and pitfall traps as sampling techniques, we found 28 ant species. One species [*Myrmica lonae*] was new to the Belgian fauna and several rare species were encountered. Three ecological groups could be distinguished based on soil preference: the first group of species was characteristic of sandy soil, the second contained species that were more numerous on peat soil [with *Sphagnum* spp.], and the third group of species had no soil preference. Ant nest numbers increased strongly between 1999 and 2000, especially in the plots that were inundated during the winter of 1999-2000, but the number of ant species did not differ significantly between years. Ant nest density showed an optimum at a Purple Moor-grass [*Molinia caerulea*] cover of about 45%; the number of species did not show such an optimum. Pitfall traps yielded more species than manual nest searching; in particular, temporary social parasites, species with a large foraging range and winged females from the surrounding habitats were missed by the latter technique. Finally, we give some recommendations for the conservation of, and suitable management measures for, ants on wet heathland.

Reprinted from Maes D., Van Dyck H., Vanreusel W. & Cortens J. [2003]. Ant communities [Hymenoptera: Formicidae] of Flemish [north Belgium] wet heathlands, a declining habitat in Europe. *European Journal of Entomology* 100: 545-555. Copyright European Journal of Entomology [2003] with permission.

Introduction

Despite the widely recognised conservation importance of wet heathlands, detailed information on the distribution and abundance of typical heathland and other species, particularly invertebrates, is scarce. Ants are among those poorly investigated invertebrates even though they play an important ecological role in many ecosystems and are increasingly used in a management and restoration context [Bisevac & Majer 1999; York 2000]. They have a major influence on soil development [especially on sites where earthworms are absent] and nutrient cycling, they often represent the largest biomass in various biotopes; and, are important predators of other arthropods [Alonso & Agosti 2000; Hölldobler & Wilson 1990; Seifert 1996]. Many heathland ant species are endangered in the few NW-European countries where their conservation status has been assessed [Falk 1991 – Great Britain; Seifert 1998 - Germany]. Dry *Calluna* heathlands have been sampled rather extensively for ants [e.g., Assing 1989; Brian 1964; Mabelis 1976], but studies dealing with wet *Erica* heathlands are rare. Furthermore, little is known about the effects of habitat degradation on ant species composition and ant nest density and few studies have examined between-year variation in the presence of species and their nest densities in the same site [Elmes *et al.* 1998].

North Atlantic wet heathland dominated by *Erica tetralix* is a semi-natural, declining habitat in Europe [Habitat 31.11 in the EU Habitat Directive 92/43/EEC]. It is restricted to a relatively narrow coastal zone with an oceanic climate from SW-Norway to Portugal [Gimmingham 1972]. The decline of heathland area in several European countries [estimated at up to 80% - Gimmingham 1981; Riecken *et al.* 1994; Webb 1989] has mainly been caused by afforestation and changes in agricultural practices [Rebane & Wynde 1997; Webb & Haskins 1980]. This in turn has lead to severe fragmentation and isolation of the remaining heathland sites and hampers the conservation of many, especially sedentary, heathland species [Webb 1989;

Webb & Hopkins 1984; Webb & Thomas 1994]. In Belgium, wet heathlands are restricted to the Campine region of NE Flanders [north Belgium] and to the 'Hautes Fagnes' region in Wallonia [south Belgium]. Wet heathland is one of the most threatened habitats in Flanders because it has declined strongly in both distribution area [85% decline - Allemeersch *et al.* 1988] and in quality [71% decline, based on the former and present 'completeness' of the habitat using indicator values of typical wet heathland plants - Van Landuyt 2002] and because its current area is very restricted [about 1800 ha] and fragmented. The high values of nitrogen deposition in Flanders [north Belgium] - on average 30-50 kg/ha/year with peaks of more than 90 kg/ha/year in some regions, Vanongeval *et al*. [1998] - cause a serious threat for the conservation and the management of heathland remnants. The nitrogen input via atmospheric deposition is now higher than what can be fixed by the heathland vegetation [5-20 kg/ha/year - Geypens *et al.* 1994; Van Gijseghem *et al.* 2000]. This nitrogen surplus, together with a lowering of the water table and lack of management measures have transformed many heathland sites into a dense and high vegetation dominated by the Purple Moor-grass *Molinia caerulea* [Aerts & Berendse 1988; Aerts *et al.* 1990; Berendse & Aerts 1984; Berendse *et al.* 1987]. The decline in habitat area and habitat quality of wet heathlands has led to a high number of typical wet heathland species being listed as threatened, e.g., the carabid beetle *Carabus clathratus* [Desender *et al.* 1995], the dolichopodid fly *Dolichopus atratus* [Pollet 2000], the butterfly *Plebeius argus* [Maes & Van Dyck 2001] and the dragonfly *Somatochlora arctica* [De Knijf & Anselin 1996].
In this article, we deal with 1] the description of ant communities on wet heathlands in Flanders, 2] fluctuations in ant nest numbers and species between two sampling years, 3] the effects of *Molinia caerulea* cover [as a measure of habitat degradation] on ant diversity and nest density, and 4] methodological differences between manual nest searching and pitfall trap sampling and their suitability for ant surveys.

Material and methods

Study areas and sampling methods

We selected 23 wet heathland [*Ericion tetralicis*, Schaminée *et al.* 1995] sites [Fig. 6.1] using the 'Biological Valuation Map' [a data base with biotopes covering the whole of Flanders, De Blust *et al.* 1994]. The extent of wet heathland in the sampled sites was derived from the Biological Valuation Map and additional GPS measurements [Table 6.1]. Ant nests were searched manually in 60 plots of 100 m^2 [10 m x 10 m] during July and August 1999 and 2000 [Table 6.1] by inspecting all possible nest sites [grass tussocks, sphagnum moss, soil, dead wood, etc. – Elmes *et al.* 1998]. Depending on the variation in vegetation structure, we spent 4-6 man-hours searching in each plot. For each ant nest we collected at least five workers in small Eppendorf tubes. In large sites we usually sampled more than one plot. Additionally, in 9 sites, eighteen plots [two per site] were sampled by means of pitfall traps [diameter = 9 cm] between 30 March 2000 and 15 March 2001. In and around each plot, six pitfall traps, filled with a 4% formaldehyde solution, were placed at a distance of about 10 m from each other and were emptied at fortnightly intervals [Parr & Chown 2001]. In the laboratory, ants were sorted out and classified using Klein *et al.* [1998] and Wardlaw *et al.* [1998] for the *Myrmica* spp. and Seifert [1996] and van Boven & Mabelis [1986] for the other species. In all plots, we determined the soil type [peat, i.e., with *Sphagnum* mosses, vs. sand] and we measured % vegetation cover in four subplots of 2 m x 2 m using the Londo scale [Schaminée *et al.* 1995]. The best represented plant species in the plots were *Molinia caerulea* [present in 98% of the plots, mean coverage 42.3%], *Erica tetralix* [92%, 25.4%], *Calluna vulgaris* [75%, 11.3%], *Gentiana pneumonanthe* [57%, 0.6%] and *Scirpus cespitosus* subsp. *germanicus* [34%, 3.7%].

Table 6.1. Plot code and plot area of the investigated sites. M1999, M2000: manual searching in 1999/2000: number of ant nests [between brackets the number of species]; P2000 = pitfall traps in 2000: number of individuals [between brackets the number of species].

Site	Plot	Area [in m^2]	M1999	M2000	P2000
1. Buitengoor-Meergoor	BUI-1-1	5126	11 [2]	-	-
	MEE-1-1	4497	5 [2]	-	-
	MEE-1-2	4497	1 [1]	-	-
	MEE-3-1	1348	8 [4]	-	-
2. Fonteintje	ZWB-2-1	52944	15 [5]	-	134 [12]
	ZWB-2-2	52944	12 [6]	-	-
	ZWB-2-3	52944		-	64 [5] -
3. Goor	GOO-1-1	1437	1 [1]	-	-
4. Groot Schietveld	GRS-1-1	7382	11 [4]	-	-
	GRS-4-1	11503	10 [1]	21 [2]	-
	GRS-5-1	2410	9 [1]	11 [2]	-
	GRS-8-1	1015	9 [5]	16 [3]	-
5. Hageven	HAG-1-1	1192	15 [4]	33 [8]	-
	HAG-15-1	816	-	18 [6]	264 [14]
	HAG-2-1	1244	10 [4]	-	-
	HAG-3-1	10574	-	19 [6]	-
	HAG-5-1	6791	19 [6]	-	190 [10]
	HAG-5-2	6791	10 [4]	-	-
	HAG-8-1	2838	19 [5]	19 [6]	-
	HAG-8-2	2838	16 [3]	-	-
6. Houthalen-Helchteren	HHH-1-1	12466	16 [6]	9 [4]	103 [12]
	HHH-3-1	2568	-	20 [5]	445 [17]
7. Kalmthoutse hei	KAL-2-1	2985	12 [2]	-	-
	KAL-3-1	10975	7 [2]	-	-
	KAL-4-1	8735	-	63 [6]	1172 [12]
8. Katershoeve	ZWB-4-1	5843	23 [5]	27 [6]	-
9. Klein schietveld	KLS-1-1	22357	-	39 [7]	-
10. Koeiven	KOE-1	14104	-	44 [5]	336 [11]
11. Korhaan	KOR-1	2274	3 [3]	-	-
12. Liereman	LIE-1-1	10288	21 [6]	-	-
	LIE-1-2	6889	12 [2]	29 [1]	-
	LIE-2-1	41563	12 [4]	-	143 [9]

	LIE-2-2	41563	-	43 [4]	-
	LIE-3-1	19593	-	29 [5]	106 [7]
13. Maten	MAT-1-1	8550	9 [3]	-	-
	MAT-2-1	692	1 [1]	-	-
14. Mathiashoeve	ZWB-1-1	17514	19 [5]	37 [7]	-
	ZWB-5-1	8634	-	50 [4]	291 [12]
	ZWB-5-2	8634	-	12 [4]	-
15. Neerharenheide	NEE-1-1	6742	13 [4]	-	-
16. Panoramaduinen	ZWB-3-1	29919	22 [5]	25 [6]	-
	ZWB-3-2	29919	36 [6]	38 [6]	266 [8]
	ZWB-6-1	2338	-	33 [6]	415 [13]
17. Slangebeekbron	SLA-2-1	547	8 [4]	-	-
18. Tenhaagdoornheide	TEN-1-1	1152	6 [3]	-	-
19. Teut	TEU-1-1	47729	6 [4]	20 [5]	117 [11]
	TEU-2-1	6250	24 [3]	-	-
	TEU-3-1	3973	20 [6]	-	85 [10]
	TEU-3-2	3973	-	12 [2]	-
20. Tielenhei	TIE-1-1	1212	3 [2]	-	-
21. Withoefse heide	WIT-1-1	25932	18 [7]	17 [6]	138 [11]
	WIT-1-2	25932	17 [5]	17 [5]	-
	WIT-1-3	25932	-	2 [1]	-
	WIT-1-4	25932	-	5 [3]	-
22. Ziepbeek	ZIE-1-1	20228	10 [4]	28 [5]	150 [6]
	ZIE-2-1	8833	29 [3]	50 [3]	-
	ZIE-3-1	10570	36 [3]	-	141 [7]
	ZIE-4-1	2274	28 [4]	-	-
23. Zwart water	ZWW-1-1	26869	11 [4]	-	333 [13]
	ZWW-1-2	26869	-	22 [4]	-
Number of plots	60		44	32	18

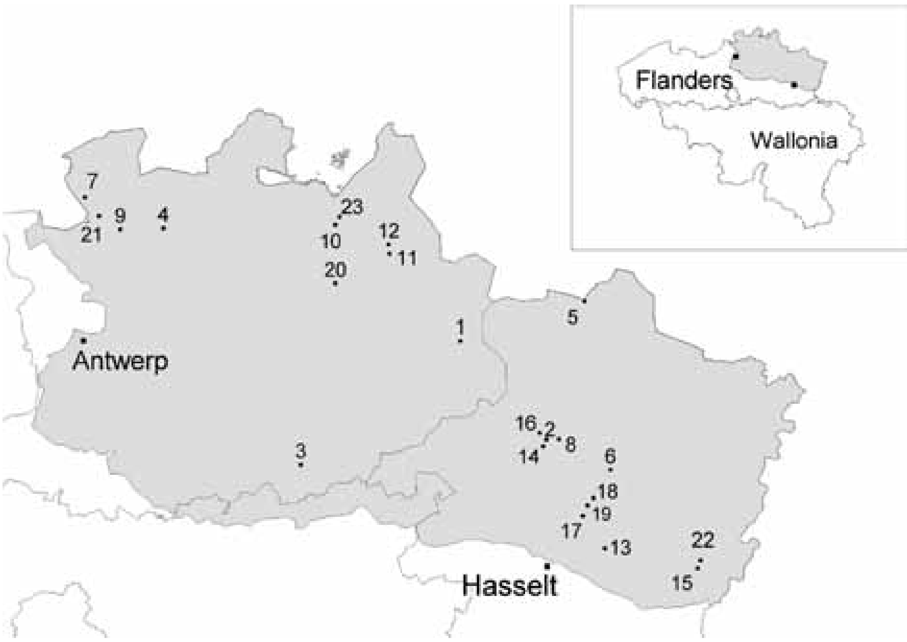

Figure 6.1. Location of the sampled sites in Belgium. The Campine region is shaded in gray both on the small map of Belgium and on the detailed map with the sampled sites. Site numbers correspond with those in Table 6.1.

Analyses

We determined ecological groups based on densities per plot using a Two Way Indicator Species Analysis [TWINSPAN - Hill 1979], using the ant data obtained from manual nest searching; if plots were sampled in both years, ant nest numbers were averaged across years; only plots with at least three ant nests [n=53] and species present in at least five plots [n=11] were used in the analysis. Differences in overall and specific ant nest densities and in species richness between 1999 and 2000 were tested using a paired t-test. The relationships between ant diversity and nest densities [averaged across years] on the one hand and % *Molinia caerulea* cover on the other was examined by a polynomial regression of the second order. Numbers of species found by manual nest searching and pitfall trap sampling were compared only for the twelve plots that were investigated by both techniques in 2000.

Results

Ant diversity and communities

During the two years of sampling, we found 28 ant species [Table 6.2], representing 53% of all indigenous species in Flanders [Dekoninck & Vankerkhoven 2001]. One species, *Myrmica lonae*, was new to the Belgian fauna.

Ant diversity tended to be positively correlated with site area [N = 47, Spearman R = 0.264, p = 0.07] but not with plant species richness [N = 52, Spearman R = -0.151, p = 0.29]; ant diversity did not differ significantly between sandy and peat soils [3.89 [n = 35] vs. 4.18 [n = 25], Kruskall-Wallis H [1,60] = 0.324, p = 0.57].
The TWINSPAN distinguished three ecological groups of ants on wet heathlands in Flanders. A first group of species was more numerous in the plots on sandy soil [Kruskall-Wallis test H [1,50] = 5.846, p = 0.016]: *Formica fusca*, *Lasius niger*, *Leptothorax acervorum*, *Myrmica sabuleti* and *Tetramorium caespitum*. A second group consisted of two species that were more abundant in the plots on peat soil [with *Sphagnum* spp.] [Kruskall-Wallis H [1,56] = 14.414, p<0.001]: *Formica transkaucasica* and *Myrmica scabrinodis*. The four remaining species did not show any preference for soil type [Kruskall-Wallis H [1,122] = 0.018, p = 0.89]: *Lasius platythorax*, *Leptothorax muscorum*, *Myrmica ruginodis* and *M. rubra*.

Table 6.2. Species list and number of nests [between brackets the number of plots in which the species was found] for all species that were found by manual nest searching in 1999 [MS1999, 44 plots] and in 2000 [MS2000, 32 plots] and the number of individuals per ant species in the pitfall traps [PF2000, 18 plots].

Species	MS1999	MS2000	PF2000
Formica cunicularia	2 [1]	3 [2]	7 [4]
Formica fusca	11 [6]	28 [9]	214 [14]
Formica pratensis	-	-	5 [5]
Formica rufa	-	-	2 [1]
Formica rufibarbis	1 [1]	-	23 [2]
Formica sanguinea	-	2 [1]	411 [9]
Formica transkaucasica	93 [14]	44 [7]	293 [11]
Lasius flavus	1 [1]	2 [1]	3 [1]
Lasius fuliginosus	-	-	15 [6]
Lasius meridionalis	-	-	17 [10]
Lasius mixtus	-	-	1 [1]
Lasius niger	34 [12]	24 [8]	444 [7]
Lasius platythorax	179 [33]	291 [25]	838 [17]
Lasius psammophilus	-	-	1 [1]
Lasius umbratus	-	-	49 [14]
Leptothorax acervorum	4 [4]	15 [5]	21 [6]
Leptothorax muscorum	2 [2]	3 [2]	21 [4]
Myrmica lonae	3 [1]	-	-
Myrmica rubra	47 [25]	82 [22]	242 [17]
Myrmica ruginodis	70 [25]	109 [23]	464 [18]
Myrmica sabuleti	24 [6]	26 [6]	823 [9]
Myrmica scabrinodis	125 [26]	203 [25]	555 [18]
Myrmica schencki	1 [1]	4 [3]	140 [8]
Stenamma debile	-	-	4 [3]
Strongylognathus testaceus	-	-	5 [2]
Tapinoma ambiguum	6 [2]	1 [1]	1 [1]
Tapinoma erraticum	1 [1]	2 [1]	12 [1]
Tetramorium caespitum	8 [3]	17 [4]	218 [5]
Number of nests/individuals	612	856	4829
Number of species	18	17	28

Nest densities and between year fluctuations in nest densities

Overall nest density varied strongly among plots [1-64/100 m^2, Table 6.1] and was positively correlated with site area [N = 47, Spearman R = 0.488, $p<0.001$]. Nest density was significantly higher in plots on peat soil [5.8 nests/100m^2, n = 25] than in plots on sandy soil [4.0 nests/100 m^2, n = 35; Kruskall-Wallis: H [1,60] = 4.901, p = 0.027], although the % cover of the most frequently used nest substrate [*Molinia caerulea* tussocks] did not differ significantly between soil types [Kruskall-Wallis H [1,51] = 0.239, p = 0.62]. The mean number of ant nests per plot was significantly lower in 1999 than in 2000, but the mean number of species per plot was similar across both years [Table 6.3]. When grouping species with different life strategies [*Lasius* spp. and *Formica* spp. with stable and long-living nests on the one hand and *Myrmica* spp. with transient nests on the other] the number of nests is only significantly different for the *Myrmica* spp. [Table 6.3]. Considering the species separately, two species had significantly higher nest densities in 2000 compared with 1999: *Myrmica ruginodis* and *Myrmica scabrinodis* [Table 6.3]. The difference between the abundance of ant nests between 1999 and 2000 can be explained by the fact that six of the sixteen investigated plots were inundated for several weeks during the winter of 1999-2000. These plots had to be re-colonised by ants in the following spring or ants had to survive inundation for several weeks. Analysing the inundated and non-inundated plots separately, showed that the number of ant nests was significantly higher in 2000 in the inundated plots [paired t-test, t = -4.259; p = 0.008] but not in the non-inundated plots [paired t-test, t = -0.847; p = 0.42]. Furthermore, the number of nests increased particularly for species that occur in wetter and cooler microclimates [*Myrmica scabrinodis*, *M. ruginodis* and *Lasius platythorax*] whereas species of drier microclimates [*Lasius niger* and *Myrmica sabuleti*] tended to decrease [Table 6.3] suggesting that microclimatic changes in the inundated plots probably caused a shift towards the part of the spectrum representing species that prefer wetter and colder conditions.

Table 6.3. Mean species and ant nest number and specific ant nest numbers in the 16 plots that were manually sampled in 1999 and in 2000 [between brackets the number of plots used in the analyses]. Differences are tested by means of a paired t-test [t]. ** = p<0.01, * = p<0.05, n.s. = not significant.

	1999	2000
# ant nests [16]	17.4	23.8**
# ant species [16]	4.3	4.5$^{n.s.}$
Species with stable nests [16]		
Lasius spp. and *Formica* spp.	8.9	11.4$^{n.s.}$
Species with transient nests [16]		
Myrmica spp.	7.6	11.7*
individual species		
Formica fusca [5]	1.2	2.2$^{n.s.}$
Formica transkaucasica [5]	7.0	7.8$^{n.s.}$
Lasius niger [5]	3.2	2.6$^{n.s.}$
Lasius platythorax [13]	6.6	9.2$^{n.s.}$
Myrmica rubra [12]	1.7	1.8$^{n.s.}$
Myrmica ruginodis [10]	3.2	4.8*
Myrmica sabuleti [4]	2.5	1.8$^{n.s.}$
Myrmica scabrinodis [14]	4.3	7.9*

The effect of *Molinia caerulea* encroachment on ant diversity and nest density

The highest overall nest densities were found on plots with a *Molinia caerulea* cover between 38-50%. Overall nest density increased non-linearly with % *Molinia caerulea* cover. A linear regression did not show a significant relationship between the variables [$R^2 = 0.001$, $F[1,48] = 0.055$, $p = 0.81$]. However, addition of the second order term for % *Molinia caerulea* cover explained a significant proportion of the variation in the overall nest density [$R^2 = 0.13$, $F[2,47] = 3.509$, $p = 0.038$]. Overall nest density reached an optimum at 40-45% *Molinia caerulea* cover [Fig. 6.2a]. Species with stable and long-lived nests [*Lasius* spp. and *Formica* spp.] show no significant linear correlation between % *Molinia caerulea* cover and nest densities [$R^2 = 0.02$, $F[1,45] = 0.932$, $p = 0.339$]; including the second order term of % *Molinia caerulea* cover did not improve the proportion of variation explained [$R^2 = 0.02$, $F[1,44] = 0.472$, $p = 0.63$]. *Myrmica* spp., with transient nests, showed a significant negative linear correlation with % *Molinia caerulea* cover [$R^2 = 0.09$, $F[1,45] = 4.626$, $p = 0.037$, Fig. 6.2b]. We found no significant [non-linear] relationship between ant diversity and % *Molinia caerulea* cover [$R^2 = 0.05$, $F[2,47] = 1.306$, $p = 0.28$].

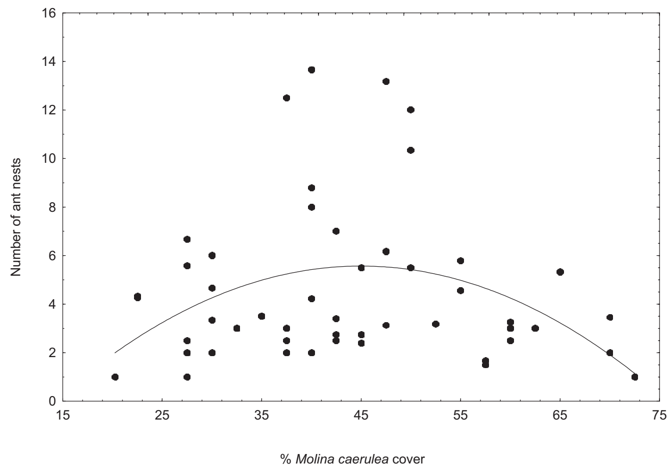

Figure 6.2a. Number of ant nests as a function of % *Molinia caerulea* cover. The line is a fit of a polynomial regression of the second order.

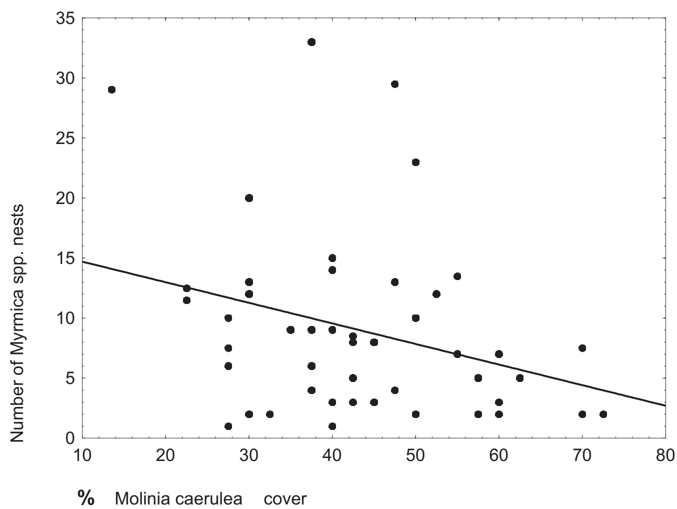

Figure 6.2b. Number of *Myrmica* ant nests as a function of % *Molinia caerulea* cover. The line is a fit of a simple regression.

Differences between manual nest searching and pitfall sampling

Pitfall traps resulted in a higher number of species than manual nest searching [Table 6.4]. If only the pitfall trap results of the fortnightly period in which the manual searching took place were compared with the results of the manual searching, six species were caught exclusively by pitfall traps: *Formica cunicularia*, *F. pratensis*, *F. rufibarbis*, *Lasius fuliginosus*, *L. meridionalis* and *L. umbratus*. The first three species are not typical of wet heathlands and may only be present at very low nest densities, which may explain their absence in the manually searched plots. The latter three species are temporary social parasites on other *Lasius* spp. [Seifert 1996] of which, almost exclusively, winged females were found; the absence of workers of the three *Lasius* spp. indicates that they do not necessarily nest in the sites in which the sexuals were found. Only one species was found by manual nest searching alone [*Leptothorax acervorum*].

The number of species per plot was significantly higher in the year-long catch from the pitfall traps [Table 6.4 - H [1,24] = 15.855; $p<0.001$], and in the pitfall traps during July and August [the two months in which the manual searching took place] [H [1,24] = 15.883; $p<0.001$] as compared with the manually searched plots. If the pitfall results of only the fortnightly period in which the manual searching took place, are used, the average number of species is no longer significantly higher in the pitfall traps [H [1,24] = 0.432; $p=0.51$].

Table 6.4. Number of plots that were both manually searched and sampled by means of pitfall traps in 2000 [total = 12] in which each ant species was found. MS = manual nest searching in 2000. PF [YC] = pitfall trap results of the complete yearly cycle [30 March 2000 – 15 March 2001]; PF [JA] = pitfall trap results of July-August 2000; PF [FN] = fortnight pitfall trap period in which the manual nest searching took place.

Species	MS	PF [YC]	PF[JA]	PF [FN]
Formica cunicularia	-	3	3	2
Formica fusca	3	10	10	5
Formica pratensis	-	5	2	2
Formica rufibarbis	-	2	2	2
Formica sanguinea	1	6	5	4
Formica transkaucasica	4	8	4	4
Lasius flavus	-	1	-	-
Lasius fuliginosus	-	5	3	2
Lasius meridionalis	-	7	7	5
Lasius mixtus	-	1	-	-
Lasius niger	4	5	5	4
Lasius platythorax	10	11	11	10
Lasius psammophilus	-	1	1	-
Lasius umbratus	-	10	10	4
Leptothorax acervorum	3	4	3	-
Leptothorax muscorum	1	2	1	1
Myrmica rubra	11	11	10	5
Myrmica ruginodis	10	12	12	7
Myrmica sabuleti	3	7	5	4
Myrmica scabrinodis	11	12	12	6
Myrmica schencki	1	4	4	3
Stenamma debile	-	2	-	-
Strongylognathus testaceus	-	1	1	-
Tetramorium caespitum	4	4	4	3
# species	13	24	21	18
Average # species per plot	5.5	11.2	9.6	6.1

DISCUSSION

Ant diversity and communities

During this study, *Myrmica lonae* was observed for the first time in Belgium [Schoeters & Vankerkhoven 2001]. It also has only recently been found in the Netherlands [Boer 1999; Elmes *et al.* 1994]. According to Wardlaw *et al.* [1998] and Elmes *et al.* [1994], *M. lonae* occurs in wetter habitats [e.g., wet heathlands] than its sister species *M. sabuleti*, although Saaristo [1995] calls *M. lonae* a species of very hot and dry places in the SW-archipelago in Finland. In Central Europe, Seifert [2000] and Czechowski *et al.* [2002] found *M. lonae* nests mainly in xerothermal habitats [e.g., dry woods and sun exposed rocky slopes] and far less in open boggy habitats [mainly in the northern part of its distribution]. We found three nests of *M. lonae* in the Liereman nature reserve in a plot with a *Molinia caerulea* cover of 60%; this corresponds better with the habitat description of Elmes *et al.* [1994; pers. comm.] and Seifert [2000] for the northern distribution range than with that of Saaristo [1995]. Other typical heathland species found during our survey were:

- *Formica transkaucasica* [a typical species of bogs and wet heathlands - Seifert 1996; van Boven & Mabelis 1986] was only known previously from a limited number of sites in Flanders [Dekoninck *et al.* 2003b; Schoeters & Vankerkhoven 2001; Vankerkhoven 1999]; we found twelve additional sites;
- *Tapinoma ambiguum* [an 'inland heathland' species – Assing, 1989; Boer, 1999] was only known from two sites in Flanders [Dekoninck *et al.* 2003; Schoeters & Vankerkhoven 2001] and was only recently found in Luxemburg [Baden 1998]; the species is very rare in Poland [Czechowski *et al.* 2002]; we found the species in two additional sites;

- *Strongylognathus testaceus* [a social parasite of *Tetramorium caespitum*, a common species of dry heathlands – Seifert 1996] was only known from two sites in Flanders [Schoeters & Vankerkhoven 2001; Vankerkhoven 1999];
- *Lasius meridionalis* [a temporary social parasite of *L. psammophilus* – Seifert, 1996] was only recently added to the Belgian fauna [Dekoninck *et al.* 2003b; Schoeters & Vankerkhoven 2001]; Both *L. meridionalis* and *L. psammophilus* are rare species in Poland [Czechowski *et al.* 2002]; we found winged females of *L. meridionalis* on nine sites and workers of *L. psammophilus* in only one site; since we only found winged females of *L. meridionalis*, we can not assume that *L. psammophilus* is present at all nine *L. meridionalis* sites as well.

Some species were absent from the wet heathlands in north Belgium [e.g., *Camponotus herculeanus*, *Formica lemani*, *Formica pressilabris*, *Myrmica lobicornis* and *Symbiomyrma karavajevi*] but are present in the same habitat type in the Hautes Fagnes in south Belgium [Bondroit 1912; van Boven 1977]. Some of these species are mountain species or boreal relics which may explain their absence in Flanders. Species richness on the studied Flemish wet heathlands is comparable with that in NW-Germany [Assing 1989] but is lower than similar Central or Eastern European habitats. Seifert [1996] mentions four additional species for wet open habitats, such as wet heathlands, for Germany that do not occur in Belgium [Dekoninck & Vankerkhoven 2001]: *Myrmica vandeli*, *M. gallienii*, *Formica uralensis* and *F. forsslundi*. However, due to a high amount of nitrogen deposition, the ant diversity in most NW-European nutrient-poor habitats [e.g., wet heathlands, bogs, species-rich grasslands] is decreasing more rapidly than less intensively cultivated areas in Eastern or Central Europa [Bobbink *et al.* 1998]. Plant diversity did not appear to be a useful surrogate for ant species richness [cf. Alonso 2000; Boomsma *et al.* 1987]. Gallé [1991] and New [2000] found the same for dunes in S-Finland and grasslands in Australia, respectively. We did find a positive correlation between area on the one hand and ant diversity and nest density on the other; if this correlation holds true for other small invertebrates [that can act as possible prey for ants], large sites may offer a larger food supply for ants and thus result in higher nest numbers. The higher species numbers in large sites may be explained by the greater variation in vegetation structure offering more possible nesting sites for a larger number of species. The correlation between both ant diversity and nest density on the one hand and area on the other, emphasizes the importance of large sites for the conservation of ants and, probably, also for other animal species.

In our classification, soil type [peat or sand] and, thus indirectly, moisture [as peat soils are wetter than sandy soils], was the main factor determining the three ecological groups. These groups correspond well with the species habitat preferences described in Brian [1964], Mabelis [1976], Assing [1989], Saaristo [1995] and Seifert [1996]. In most ant studies on heathlands, moisture and vegetation structure are the most important factors separating ant communities [Boomsma & de Vries 1980; Brian 1964; Elmes & Wardlaw 1982; Gallé 1991]. Given the limited extent of the studied region, macroclimatic differences are not very likely to have influenced ant distribution in the Campine region [an area of about 100 x 50 km]; microclimatic data are not available for the different study sites.

Nest densities and between year fluctuations of nest densities

Nest density was higher on peat soils than on sandy soils and large sites had higher nest densities. The higher nest density on peat soil was caused by species like *Myrmica scabrinodis* and *Formica transkaucasica*. Since the studied sites on peat soil are significantly larger than those on sandy soil [Kruskall-Wallis H [1,46] = 5.363, p = 0.021], the higher mean nest densities on larger areas might be explained by soil type and not necessarily by area. The number of ant species did not differ significantly between 1999 and 2000. However, the number of nests was significantly higher in 2000 than in 1999. Sampling itself was most probably not responsible for the differences in densities or species turn-over across the years [because the same people performed the manual searching], but could have caused ant colonies to move among nesting sites. Clearly more research is needed on this subject focusing on the repeated sampling of plots during the year, observations of the ants behaviour after sampling, etc. [Elmes *et al.* 1998]. The generalized statement of Steiner & Schlick-Steiner [2002] that ant nests are very sedentary and that their densities do not vary much between years is not supported by our data. Differences across both years were only significant on the inundated plots where a much larger number of nests was found. As described by Boomsma & de Vries [1980] for *Lasius niger*, *Myrmica rubra* and *M. scabrinodis*, ants can survive inundations of 2-14 weeks using oxygen that is stored in and between roots and litter. The inundated plots had a more open structure and a wetter microclimate in the spring following inundation offering more suitable nesting sites for species of cooler and wetter habitats such as *Myrmica scabrinodis*, *M. ruginodis* and *Lasius platythorax*.

The effect of *Molinia caerulea* encroachment on ants

Molinia caerulea tussocks are by far the most frequently used nest substrates in the wet heathland sites we studied. However, sites with a very high *Molinia caerulea* cover are expected to have a cooler microclimate at ground surface level [Bobbink *et al.* 1998; Van Dyck, pers. obs.] which reduces the invertebrate diversity [Thomas *et al.* 1999] in general, and the potential number of sunlit, warm nesting sites for ants in particular [Elmes & Wardlaw 1982; Thomas 1995; Thomas *et al.* 1998d]. de Boer [1978] found a significantly higher number of ant nests in wet *Erica* heathland with a relative high *Molinia caerulea* cover [on average 30%] than in sites with lower *Molinia caerulea* densities [on average 13%]. When vegetation cover of *Molinia caerulea* tussocks became too dense, de Boer [1978] observed negative effects on ant nest densities. Our results confirm that there is indeed an optimal % *Molinia caerulea* cover for nest density [about 40-45%]; a further increase in the % *Molinia caerulea* however resulted in a lower number of nests. The higher nitrogen deposition in the last few decades [Bobbink *et al*. 1992; Vanongeval *et al*. 1998] has caused a very strong increase in *Molinia caerulea* cover in oligotrophic habitats such as wet and dry heathlands [Bobbink *et al*. 1998; Chambers *et al*. 1999]. More research is needed to investigate the impact of degradation of wet heathlands on invertebrates and other faunal elements in general and on ant colony sizes in particular [cf. Bobbink *et al*. 1998; Elmes & Wardlaw 1982].

Despite their important role in most ecosystems and their potential as bioindicators [cf. York 2000], invertebrates in general and ants in particular are seldom used in nature management and restoration evaluation. Most management and restoration measures in heathlands are based mainly on plant diversity [e.g., Jansen *et al*. 1996; Smith *et al*. 1991]. Plants represent only a small fraction of the biodiversity present in ecosystems [Thomas 1994]. Differences in scale, habitat use and mobility call for specific management measures for invertebrates that are not met by using only birds, mammals or vascular plants as target species [Webb & Thomas 1994]. Much more research is needed to estimate the impact of management and restoration measures on ant communities on wet heathlands [e.g., long-term monitoring of different management practices, rate of colonisation on restored heathlands, influence of the neighbouring unmanaged habitats on rate of colonisation, etc.]. Management measures [e.g., large-scale sod cutting and burning] can be very beneficial for the restoration of wet *Erica* heathland vegetation [Jansen *et al*. 1996], but can cause severe damage to ant communities and nesting sites. Gorssen [1999] found that sod cut and burned plots had a lower number of nests and species than unmanaged heathland plots. Brian *et al*. [1976] and de Boer [1978] did not find any differences in ant species composition and even found higher nest numbers after burning of dry heathland. The very high values of nitrogen deposition however, suggest that burning is at present no longer recommended because fast growing grasses such as the Purple Moor-grass, *Molinia caerulea*, will become dominant and reduce nest densities. Low intensity grazing and mowing also reduced the number of nests compared with unmanaged plots but seem to be less detrimental than burning and sod cutting [de Boer 1978; Mabelis 1976]. Restoration of heathland sites on former mining grounds in Australia showed that 20 years after the rehabilitation of the mining grounds, the original ant assemblage structure still had not been achieved [Bisevac & Majer 1999]. Following York [2000], we recommend low-intensity and small-scaled management and restoration measures on degraded wet heathland sites to minimise the effects on the existing ant diversity and its associated [myrmecophilous] communities. Management measures are not only necessary in degraded wet heathland sites even high quality wet heathlands need regular small-scale management to maintain and/or create suitable nesting sites for ants and other invertebrates. Invertebrates can thus be used in a complementary way to other, more frequently used biota [e.g., vascular plants, birds] in managing or restoring degraded sites [Thomas 1994].

Differences between manual nest searching and pitfall sampling

Combining different trapping techniques [pitfall traps, manual searching, or litter extraction] gives the most complete information on ant species richness and densities [Andersen 1997; Delabie *et al*. 2000; Parr & Chown

2001; York 2000]. However, the information needed will determine which [combination of] sampling techniques is the most cost-effective one [Bestelmeyer *et al*. 2000]. If only a species list of a relatively large number of wet heathland sites is needed, our data showed that three pitfall traps are sufficient [cf. Stein, 1965; Kabacik-Wasylik, 1970]. On average 81% of the total species richness per site was caught in one of the three pitfall traps, while the additional pitfall traps only added 14% and 4% respectively to the total species richness per site. When sampling efforts must be limited in time, pitfall traps can best be placed between mid July and mid August: 20 of the 28 species were caught during this period and the remaining eight species were only found in very small numbers [except for *Tapinoma erraticum* that was mainly caught in spring]. Advantages of pitfalls are the possibility of sampling many sites simultaneously [Bestelmeyer *et al*. 2000; Greenslade 1973; Parr & Chown 2001] and the ability to find social parasites and ants with hidden nests [especially winged females during the mating season]. Disadvantages of pitfall traps are the large number of individuals to be classified, the lack of information on nest densities and the fact that the numbers of individuals per ant species can not be compared between species and sites due to different activity patterns and differences in catching ratio [i.e., the number of ants finally caught against the total number of trap contacts – Bestelmeyer *et al*. 2000; Seifert 1990]. Disadvantages of manual nest searching are its very time consuming nature [about 4-6 man-hours per 100 m^2], the disturbance caused to the nests, and the difficulty of finding social parasites [Bestelmeyer *et al*. 2000].

7 Conservation of the threatened Alcon Blue butterfly *Maculinea alcon* in Belgium

FOTO: JEROEN MENTENS

"... the need to base conservation on detailed ecological research .... The failure to save British Maculinea arion populations, after five reserves had been established and after 50 years of expensive attempts based on educated guesses, is just one example of the false economy of omitting this vital first step. Species can be saved from the brink of extinction if the knowledge exists to do this, but eleventh hour research is uneconomical: the work is easier, quicker, cheaper, and more likely to result in success if species are studied earlier in their declines."

Jeremy Thomas [1991]. Rare species conservation: case studies of European butterflies. In The scientific management of temperate communities for conservation [Spellerberg I.F., Goldsmith F.B. & Morris M.G. eds]. Blackwell Scientific Publications, Oxford. pp. 149-197.

To organize and prioritise species-specific conservation efforts, we delineate 'functional conservation units' for the threatened Alcon Blue butterfly *Maculinea alcon* in Belgium. We used detailed distribution data on the butterfly, its host plant and its habitat, present-day population sizes and its mobility and colonization capacity to determine functional conservation units on different spatial scales: FCU-1, i.e., the twelve presently occupied habitat patches plus the area within a range of 500 m surrounding them [the maximum local movement distance, based on mark-release-recapture data], FCU-2, i.e., the areas within a range of 2 km around the occupied habitat patches [the maximum observed colonization capacity] and FCU-3, i.e., potential re-introduction sites [sites where *M. alcon* went extinct recently]. We suggest different management and planning measures for each type of functional conservation unit and discuss translocation and re-introduction as 'intensive care' conservation measures for this threatened and sedentary species.

Reprinted from Maes D., Vanreusel W., Talloen W. & Van Dyck H. [in press]. Functional conservation units for the endangered Alcon Blue butterfly *Maculinea alcon* in Belgium [Lepidoptera, Lycaenidae]. *Biological Conservation*.

Introduction

In an era where habitat fragmentation and habitat destruction are causing declines and local extinctions of many species, restoring local or regional habitat networks for target species has become an important conservation strategy throughout the world [e.g., Amato *et al.* 1995; Cowley *et al.* 2000; Poiani *et al.* 2000; Bergman & Landin 2002]. Both policy makers and field conservationists need to take decisions on where and how to implement species-specific conservation measures in addition to more general area- or biotope-oriented conservation. Decision-making tools based on biologically relevant – in this case species-specific – knowledge can help maximizing the chances on success of these measures. For instance, the probability of a successful colonisation of restored habitat by a target species is affected by dispersal capacity, the spatial configuration of habitat and the size of source populations. Whether two populations belong to the same [future] network or should be regarded as isolated ones, depends on the mobility of the target species and on the nature of the intervening matrix [Ricketts 2001; Keyghobadi *et al.* 2003]. Moreover, habitat has often been treated too vaguely as vegetation types, but requires more careful definitions in terms of essential resources for the conservation of butterflies [among many other taxa] [Dennis *et al.* 2003].

In case of threatened species, conservation management should anticipate on species requirements at different spatial levels ranging from local habitat quality to habitat network geometry at the landscape level. In highly deteriorated landscapes, conservation efforts should not only be limited to sites where target species occur, but should also be expanded to sites with high potentials for the target species. Therefore, the recognition of clearly defined spatial conservation units with an associated program of measures for each level can be a useful tool to help guiding the conservation process. In order to base such a tool on solid scientific knowledge, detailed knowledge on the distribution, dispersal and colonization capacities and habitat requirements of the focal species are required.

In the case of threatened species with a limited number of remnant populations in a particular focal region, spatial risk spreading strategies may contribute significantly to bridge the critical time lag between habitat restoration measures and their effects on habitat quality and quantity. Risk spreading can include translocations to suitable, unoccupied sites that have a low probability of spontaneous colonization on the short term or reintroductions into previously occupied sites [Oates 1992]. Such labour and knowledge intensive – and hence expensive – approaches have to be seen as 'intensive care conservation' rather than maintenance management. But, especially in countries with a high pressure on biodiversity like in Belgium, such measures will be temporarily necessary to preserve small populations of threatened species [e.g., Maes & Van Dyck 2001]. However, conservation agencies seem to be reticent on translocation and reintroduction and often lack official policies to deal with these options. Hence, translocations and reintroductions have sometimes been executed secretly which hampers insights on the colonization capacity of species. Here, we discuss the use of reintroduction and translocation within the framework of functional conservation units.

Since the 1950's, butterfly diversity decreased severely in Belgium and urgent measures are needed to preserve several remaining threatened species [Maes & Van Dyck 2001]. The most important factors for the decline in butterfly diversity are biotope loss, fragmentation of habitats in biotope remnants, and declining habitat quality, especially in wet and nutrient poor biotopes [Maes & Van Dyck 2001]. In particular, wet heathlands and bogs have strongly degraded both in area and quality. The reduction in area is estimated to be > 85% in Flanders [Allemeersch *et al.* 1988]. Biotope quality declined with 71% [estimate based on 'completeness' using indicator values of typical wet heathland plants - Van Landuyt 2002]. In Belgium, but also throughout Europe [cf. EU Habitat Directive], wet heathlands are of

high conservation value [Rebane & Wynde 1997; Webb 1998]. One of the most typical butterfly species of wet heathlands in Belgium is the Alcon Blue butterfly *Maculinea alcon* [Denis & Schiffermüller 1775] that is a conservation target both in Europe [Munguira & Martín 1999] and in Belgium [Vanreusel *et al.* 2000]. Several authors have stated that *M. alcon* is able to survive in small habitat units [<1 ha], even with low host plant densities as long as suitable host ants are present [Tax 1989; Bink 1992; Wynhoff 1996]. The rationale behind this is that the butterfly's only host plant [*G. pneumonanthe*] is perennial [up to 30 years] and responds very slowly to environmental changes [e.g., desiccation, eutrophication, etc.]; therefore, adult, flowering individuals can survive for relatively long times in vegetations that no longer allow recruitment [Oostermeijer *et al.* 1992]. This time lag between habitat deterioration and decline of the species may mislead managers who only rely on presence/absence data of the flowering host plant and of the butterfly. Small population sizes and/or small patch sizes of *G. pneumonanthe* both affect the population structure due to genetic bottlenecks and have negative effects on seed setting and rejuvenation [Oostermeijer *et al.* 1998]. Furthermore, environmental influences that affect population structure [through negative effects on germination] have a higher impact in small areas [Vanreusel & Smets 2002].
As it is the case elsewhere, budgets for conservation [particularly for species conservation] are limited in Belgium, and an adequate conservation relies on clear goals, programs and underpinned priorities on the one hand and on a good co-operation between ecologists, managers and policy makers on the other [Wilson & Lantz 2000]. In this article, we define functional conservation units on different spatial scales in order to help organizing and prioritising species-specific conservation efforts for *M. alcon* in Belgium. The delineation of these units are validated with data on i] distribution [including detailed measurements of habitat patches] and changes

in distribution of the butterfly, its host plant and habitat, ii] population sizes [based on egg counts] and iii] mobility and colonization capacity [based on mark-release-recapture data and recolonization events]. These units are used to rank the priority of species-specific measures. The optimal scale and choice of conservation measures [including their intensity] differs among the units. Finally, we discuss translocation and re-introduction as 'intensive care' conservation measures for this threatened and very sedentary species.

Methods

Study species and study sites

M. alcon is an obligate ant parasite butterfly with a scattered distribution in Europe [Wynhoff 1998b]. The Marsh Gentian *Gentiana pneumonanthe* is its single host plant in Belgium [Maes & Van Dyck 1999] and different *Myrmica* ants are used as host ants throughout Europe [Thomas *et al.* 1989; Elmes *et al.* 1994]. Apart from some doubtful records in western and southern Belgium, *M. alcon* has always been restricted to wet heathlands with *Erica tetralix*, bogs and nutrient poor hay meadows in the Campine region [NE Belgium, Fig. 7.1; Maes & Van Dyck 1999; Goffart & De Bast 2000]. Its host plant declined in distribution area by at least 64% in the last 30 years [Biesbrouck *et al.* 2001]. The three potential host ant species *Myrmica ruginodis*, *M. rubra* and *M. scabrinodis* [Elmes *et al.* 1994] are, however, rather common in Flanders [Schoeters & Vankerkhoven 2001]. Detailed historical distribution data are not available for ants in Belgium, making estimates of changes in distribution of the host ants impossible.
In 1999 and 2000 we investigated 39 wet heathland sites in the Campine region where both wet *Erica tetralix* heathland [data from Biological Valuation Map; De Blust *et al.* 1994] and *G. pneumonanthe* were present

[data from Florabank; Biesbrouck *et al.* 2001]. These included all present and formerly known sites of *M. alcon* in Belgium. Table 7.1 gives the conservation status and the area of wet heathlands in the investigated sites. Typical dominant plant species in the study sites were Purple Moor-grass [*Molinia caerulea*, average coverage 42%], Cross-leaved Heath [*Erica tetralix*, 24%], Heather [*Calluna vulgaris*, 9%] and Deer grass [*Scirpus cespitosus* subsp. *germanicus*, 4%].

Mark-Release-Recapture [MRR] and colonization events

In 1997, we carried out MRR-studies in the nature reserves of Liereman [Oud-Turnhout, N 51°20 E 5°05] and Zwarte Beek [Koersel-Beringen, N 51°05' E 5°20'], where we studied two different populations [Panoramaduinen and Fonteintje] that are separated by about 1 km of woodland and meadows [Fig. 7.1]. *M. alcon* individuals were caught by hand net, marked with a unique number on the ventral left hind wing with a permanent marker and released on the spot of capture. Distances between consecutive capture points were measured by theodolite in Liereman and by hand meter in Zwarte Beek. Maximal distances between the outer boundaries in each of the three populations were 650 m, 275 m and 410 m in Liereman, Panoramaduinen en Fonteintje respectively.
We estimated the colonization ability of *M. alcon* from i] occasional observations of adult butterflies away from permanently occupied habitat patches and ii] observations of *M. alcon* eggs on *G. pneumonanthe* in habitat patches that were previously unoccupied and hence colonised during the year of observation. In addition, we observed the behaviour of a small subsample of *M. alcon* males released in non-habitat [a woodland ride and an improved grassland].

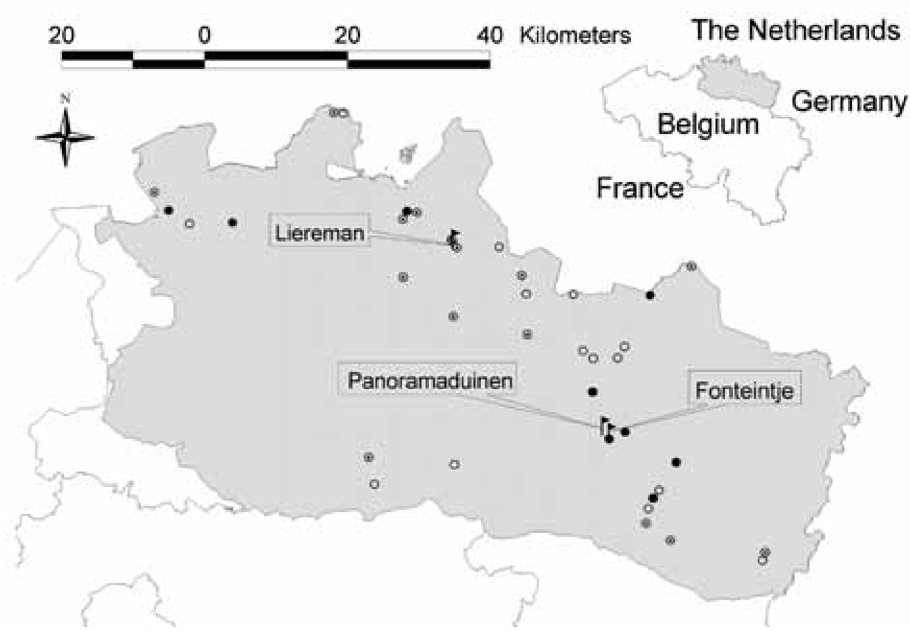

Figure 7.1. Location of the investigated sites; sites with present-day populations of *M. alcon* are marked with black dots [sites where the MRR-study was performed are marked with flags]; sites were *M. alcon* went extinct are marked with dotted circles and wet heathlands where *M. alcon* has never been documented are marked with an empty circle. The Campine region is shown in grey.

Distribution and habitat use

Potential habitat patches for *M. alcon* were determined as wet *Erica tetralix* heathlands with *G. pneumonanthe* populations and with *Myrmica* spp. ant nests. The size of the patches was determined by the outer limits of *G. pneumonanthe* populations. The habitat patches were localised and measured with a global positioning system [GPS] corrected by a base station [precision 1 m]. In all sites, we counted the number of *G. pneumonanthe* plants and, if the butterfly was present, all *M. alcon* eggs, except for one site [Fonteintje] where, due to the very large number of plants, only about 1/3 of the *G. pneumonanthe* plants was counted. The white eggs are very conspicuous on the green flower buds of *G. pneumonanthe*; caterpillars hatch through the basal side of the egg [Thomas *et al.* 1991] and most of the [empty] egg shells remain on the host plant until about two weeks after the flight season [Ebert & Rennwald 1993a]. We estimated the number of adult butterflies in each population by assuming that every female lays on average 50-100 eggs and that the sex ratio is 1, based on other *Maculinea* species [Hochberg *et al.* 1992, 1994; Meyer-Hozak 2000; Griebeler & Seitz 2002]. We searched host ant nests by inspecting all possible nest substrates in 62 plots of 10 x 10 m^2 in 24 of the 39 investigated sites [Maes *et al.* 2003]. In order to test for differences in plant species cover [especially *Molinia caerulea* cover; Berendse & Aerts 1984] between present-day populations and sites where populations went extinct, we estimated plant species cover in all sites in four subplots of 2 x 2 m^2 within a plot of 10 x 10 m^2 using the Londo scale [Londo 1976].

Statistical analyses

We analysed the spatial patterns of occupied and vacant flight areas with a logistic regression with presence/absence as dependent variable and flight area and distance to the nearest population [both $\log_{10}$-transformed to obtain normality] as independent variable. For the calculation of distances between two consecutive captures, we only used the recaptures with at least one day time interval. Differences in distances moved were analysed by means of a 2-way ANOVA with site and sex as independent variables and distance [$\log_{10}$-transformed to obtain normality] as dependent variable. We used a logistic regression to detect differences in plant cover between sites with and without *M. alcon*. Subplots were grouped per 10 x 10 m^2 plot. All analyses were done with the Statistica software package [StatSoft Inc. 2001].

Table 7.1. Status of present-day and extinct populations of *M. alcon* in Belgium with information on the ownership [M = Military area, N = non-governmental nature reserve, F = Flemish nature reserve, P = private property, C = city property]; the area of wet heathland in ha according to the Biological Valuation Map [WH], the total area of the patch and the number of separate habitat patches [FA [#P]], the number of *Gentiana pneumonanthe* in the habitat patch [#GP], the density of the three potential host ant nests per 100 m^2 [Dens.HA]: rug = Myrmica ruginodis, rub = Myrmica rubra and sca = Myrmica scabrinodis. EPS = population size based on the number of eggs: very small = < 100 adults, small = 100-400 adults, large = > 400 adults.

Site	Status	WH [ha]	FA [#P] [ha]	#GP	Dens.HA rug	rub	sca	#eggs [EPS]
Current populations [site codes in Fig. 7.3]								
1. Groot Schietveld [GRS]	M	401.6	>10.3 [>7]	>1646*	0.2	0.3	2.3	>2975 [small]
2. Hageven [HAG]	N	15.3	3.0 [8]	1662	3.0	2.4	3.3	4431 [small]
3. Liereman [LIE]	N	53.1	4.4 [6]	515	3.9	2.4	2.0	5506 [small]
4. Sonnisheide [HHH]	M	?	1.3 [1]	871	5.5	1.0	0.5	4611 [small]
5. Teut [TEU]	F	48.9	4.8 [1]	242	6.0	0.5	4.5	5472 [small]
6. Visbedden [VIS]	M	136.3	1.3 [1?]	.	.	.	.	. [?]
7. Withoefse heide [WIT]†	F	16.1	2.7 [1]	44	3.5	2.0	0.3	456 [very small]
8. Zwarte Beek		133.9						
8a. Mathiashoeven [ZWB-1]	M		1.8 [1]	172	4.5	4.0	11.5	4 873 [small]
8b. Fonteintje [ZWB-2]	M		5.3 [2]	>426*	2.5	1.6	1.8	>12798 [large]
8c. Panoramaduinen [ZWB-3]	M		3.0 [1]	114	3.8	2.8	5.8	3510 [small]
8d. Katershoeve [ZWB-4]	M		1.3 [6]	380	4.5	1.5	7.5	1843 [very small]
9. Zwart Water [ZWW]	N	16.4	3.3 [2]	491	2.0	0.5	1.0	2287 [very small]
* = only part of the total population was counted, † = the population went extinct in 2001								
Extinct populations [year of extinction]								
10. Buitengoor [1998] [BUI-MEE]	N	42.9	1.4	10-20	-	1.7	2.0	-
11. Goor [1998] [GOO]	N	0	0.1	1-5	-	-	-	-
12. Wolfsven [1998] [WOL]	F	2.1	0.03	1-5	.	.	.	-
13. Ziepbeek [1998] [ZIE]	F	92.3	2.1	50-100	2.3	0.3	15.2	-
14. Tielenhei [1997] [TIE]	M	0	0.2	10-20	2.0	-	-	-
15. 's Gravendel [1995] [GRA]	P	0	0.3	1-5	.	.	.	-
16. Zwarte heide [1995] [ZWH]	N	1.2	0.6	50-100	.	.	.	-
17. Kauwbosstraat [1994] [KAU]	C	0	0.2	10-20	3.5	4.0	2.0	-
18. Korhaan [1994] [KOR]	N	2.1	0.2	1-5	1.0	-	1.0	-

Site	Status	WH [ha]	FA [#P] [ha]	#GP	Dens.HA rug	rub	sca	#eggs [EPS]
19. Kalmthoutse heide [1993] [KAL]	F	281.5	0.3	50-100	6.0	-	0.7	-
20. De Maten [1973] [MAT]	N	22.9	0.9	10-20	-	2.5	-	-
21. Ronde Put [1973] [RON]	F	9.7	1.3	1-5	.	.	.	-
22. Hei van Van Damme [1970-79] [DAM]	N	0	0.1	1-5	.	.	.	-
23. Hoge Mierdse Hei [1970-79] [HMH]	N	0	0.02	1-5	.	.	.	-
24. Koeiven [<1970] [KOE]	P	2.0	1.4	-	12.0	2.0	-	-
25. Meerseldreef [1947] [DRE]	N	0	0.9	1-5	.	.	.	-
Wet heathland sites with *Gentiana pneumonanthe* where *M. alcon* has never been documented								
26. Elsakker	F	3.2	-	1-5	.	.	.	
27. Gerhagen	F	3.8	-	1-5	.	.	.	
28. Goorken	P	2.8	-	10-20	.	.	.	
29. Kattenbosserheide	N	0	-	-	.	.	.	
30. Klein Schietveld	M	73.5	-	1-5	4.0	2.0	4.0	
31. Koemook	P	0	-	1-5	.	.	.	
32. Langdonken	N	0	-	10-20	.	.	.	
33. Moensweyer	F	1.2	-	-	.	.	.	
34. Neerharenheide	F	33.5	-	10-20	1.0	-	2.0	
35. Plat-Holven	N	4.3	-	-	.	.	.	
36. Riebos	N	1.7	-	10-20	.	.	.	
37. Slangebeekbron	N	8.7	-	10-20	1.0	-	-	
38. Tenhaagdoornheide	F	59.8	-	10-20	-	2.0	-	
39. Vriesput	M	0	-	1-5	.	.	.	

Results

Movements and colonization

Table 7.2 gives an overview of the results of the MRR-study. In total, we caught 576 individuals in the three populations. In Liereman, the recapture ratio did not differ between males and females. In both populations of Zwarte Beek the recapture ratio was significantly higher for males. The overall recapture ratio [34%] however did not differ significantly between sexes [Table 7.2]. The overall mean movement of males and females differed among sites resulting in a significant two-way interaction [Table 7.2]; both in Fonteintje and in Panoramaduinen males moved longer distances than females, while in Liereman the opposite was true. The maximum recorded distance moved was larger in females than in males in Liereman and in Panoramaduinen, but shorter in Fonteintje [Table 7.2]. The majority of the individuals was very sedentary: 63% of the males and 71% of the females moved less than 50 m between two consecutive captures; only a small proportion of all recaptured individuals covered distances larger than 150 m [7% for both males and females, Fig. 7.2]. In Zwarte Beek, we did not observe movements of individuals between the two investigated populations.

The data on colonization events of empty habitat patches [Table 7.3] indicate that dispersal distances can be much longer than the maximum distances recorded in MRR-studies. The observation of 100 *M. alcon* eggs [probably coming from one or two females] at almost 7 km from the nearest known population, is most probably the result of a 'secret' re-introduction [Ghis Palmans, pers. comm.]. This re-introduction was unsuccessful since no more eggs were found in the following years.

Observations of behaviour at edges of habitat patches indicated that *M.*

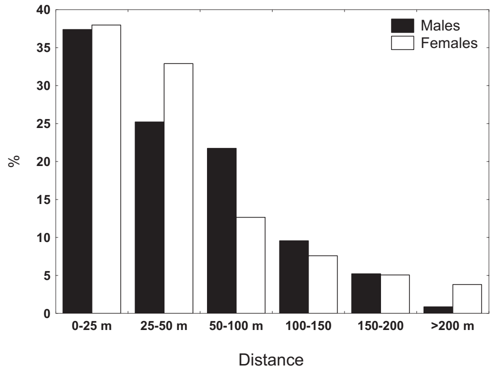

Figure 7.2. Frequency distribution of distances moved by males and females of *M. alcon*.

Table 7.2. Movement statistics from the MRR-study of *Maculinea alcon* in three study sites in NE Belgium. Differences between sexes in the numbers marked and recaptures were tested using X^2-test; overall differences between sexes and sites in moved distances were tested using a two-way ANOVA.

	N marked	N recaptured	N recapture events	Mean distance [m]	Longest single move [m]	Longest cumulative move [m]
Liereman						
Males	125	36	45	33 ± 32	114	235
Females	116	38	42	68 ± 108	500	509
Panoramaduinen						
Males	51	23	22	46 ± 35	149	263
Females	37	11	11	36 ± 52	190	206
Fonteintje						
Males	148	60	48	76 ± 57	221	409
Females	97	30	25	55 ± 56	193	229
Overall						
Males	324	119	115	53 ± 49	221	409
Females	252	79	78	59 ± 88	500	509
	p=0.033	p=0.23		F[sex] = 1.418; p=0.24 F[site] = 2.775; p=0.07 F[interaction] = 4.868; p=0.009		

alcon mostly returns to the patch when it encounters woodland edges. The few release experiments in a potential corridor [large woodland ride nearby a flight area on wet heathland, n = 5] showed that individuals flew straight upwards, leaving the ride by flying over the trees [c. 8 m height] instead of flying along the ride as we originally expected; the released males in non-habitat [improved meadow] showed a zigzag searching flight behaviour before alighting on available nectar sources that are absent on typical heathlands [*Taraxacum* sp. and *Trifolium* sp.]; afterwards, they left the meadow by flying straight over the adjacent woodland. Although adults mostly fly close to the vegetation at low speed, one adult in Fonteintje was seen passing a dense *Molinia caerulea* vegetation at a height of 3-4 m in a straight line at high speed. Although based on small sample sizes, these observations clearly indicate different behavioural patterns in habitat and non-habitat conditions.

Table 7.3. Minimal distances between newly colonised habitat patches and the nearest known population of *M. alcon* in Belgium observed between 1999 and 2001. A colonization event was determined by observing adult butterflies or eggs in a site that was unoccupied in the previous years. * = suspicion of a secret re-introduction instead of natural colonization.

Site	Distance [m]
Fonteintje	165
Katershoeve	595
Teut	650
Plateaux [NL]	700
Liereman	835
Teut	940
Plateaux [NL]	1700
Riebos*	7000

Distribution and habitat use

M. alcon declined in distribution area from 39 UTM [Universal Transverse Mercator] grid squares [5 x 5 km] in the period 1901-1950, over 24 grid squares between 1951-1970 and 18 grid squares between 1971-1990 to 12 grid squares in 1999-2000. One of the present-day populations concerns a private re-introduction after extinction in 1995 [Vanreusel *et al.* 2000]. Using grid squares as units for the trend calculation, *M. alcon* showed a decline in distribution area of 70% in Belgium in the 20th century [Maes & Van Dyck 2001] which is most probably an underestimate [Thomas & Abery 1995; León-Cortés *et al.* 1999; León-Cortés *et al.* 2000]. Using sites instead of grid cells, present-day populations of *M. alcon* can be found in nine sites. Since 1999 the species went extinct in at least 16 sites. Most of the sites have one or a few habitat patches with one [meta]population. Considering flight areas separated by at least 500 m of non-habitat as populations, the actual number of *M. alcon* populations in Belgium is reduced to 12 [Fig. 7.1].
The total area of *M. alcon* sites in Belgium in the period 1999-2000 was 42.4 ha [i.e., 0.02% of all wet heathlands in Belgium]. The spatial pattern of vacant [N=17] and occupied sites [N=11, the re-introduced population was considered extinct] showed that the probability of a patch being occupied increased with habitat patch size and decreased with distance to the nearest occupied patch [Fig. 7.3]. Populations that went extinct in the last decade were mainly located in small habitat patches and the few larger sites where the species went extinct [e.g., Ziepbeek, Buitengoor] were isolated ones [≥ 10 km away from the nearest population – Fig. 7.3]. The mean nearest neighbour distance for all present-day populations is 6.2 km [range = 0.8 – 18.3 km].
The estimated population sizes are given in Table 7.1: only one population can be considered as large in Belgium [> 400 adult butterflies], while all others are very small to small [< 400 adult butterflies].

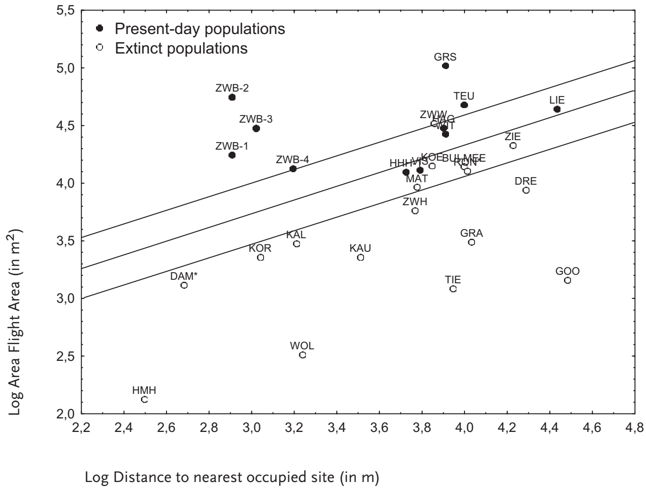

Figure 7.3. Distribution of occupied and extinct sites in relation to flight area and distance to the nearest population [site abbreviations are given in Table 7.1]. Lines indicate the probability [90%, 50% and 10%] of the presence of *M. alcon*. Logistic regression: χ^2=25.842, df=2, p<0.001; parameter estimate for $\log_{10}$area [m^2] = 10.366 and for $\log_{10}$distance [m]= -7.260.

Description of present-day *M. alcon* populations

Table 7.1 indicates that most of the current Belgian *M. alcon* populations are small and are located on a very limited area. Although the number of eggs may seem fairly high in some populations [e.g., Fonteintje], the actual number of butterflies does not exceed 1 000 individuals in 11 out of 12 populations. A logistic regression analysis did not detect a significant difference in plant cover between present-day and former *M. alcon* sites [χ^2 [15]=22.44; p>0.10]. Differences between present-day populations and extinct ones were the larger area of wet heathland in which the habitat patch was situated, larger habitat patch areas and a higher *G. pneumonanthe* cover [cf. Wallis de Vries 2004]. Host ant densities did not differ between present and former populations. Seven of the current Belgian *M. alcon* populations are located in military areas [Table 7.1] and all present-day populations are in areas under protection of the European Habitat Directive and/or Bird

Directive. Most of the populations are either managed by the Ministry of Nature Conservation [including some of the military areas] or by non-governmental nature conservation organisations.

Management measures applied in the current populations are summed up in Table 7.4. Seven sites are grazed by either horses, cattle, or sheep [or a combination of these grazers]. In the majority of the sites, sod-cutting is used as a management measure to create suitable germination sites for the host plant *G. pneumonanthe*. At present, only at one site exclosures are used to reduce grazing pressure in host plant areas.

Table 7.4. Management measures in the current *M. alcon* populations in Belgium. Gr = Grazing [H = horses, C = cattle, S = sheep]; Co = combing [removing decaying litter from *Molinia caerulea* tussocks]; Exc = exclosure [excluding grazers from the most dense *G. pneumonanthe* patches]. SC = sod cutting; Mw = mowing; Bu = burning; Ch = choppering [creating open ground by mowing into the ground with a brushcutter]; Manager: MNC = Ministry of Nature Conservation, NGNO = non-governmental nature organisation, MA = military authorities.

Site	Gr	Co	Exc	SC	Mw	Bu	Ch	Manager
1. Groot Schietveld [GRS]	-	-	-	-	-	+	-	MNC
2. Hageven [HAG]	HC	+	+	+	+	-	+	NGNO
3. Liereman [LIE]	H	-	-	+	-	-	-	NGNO
4. Sonnisheide [HHH]	-	-	-	-	-	-	-	MA
5. Teut [TEU]	-	-	-	-	-	-	-	MNC
6. Visbedden [VIS]	-	-	-	-	-	-	-	MA
7. Withoefse heide [WIT]†	-	-	-	+	-	-	-	MNC
8. Zwarte Beek								
8a. Mathiashoeven [ZWB-1]	C	-	-	+	-	-	-	NGNO/MNC
8b. Fonteintje [ZWB-2]	S	-	-	+	-	-	-	NGNO/MNC
8c. Panoramaduinen [ZWB-3]	S	-	-	+	+	-	-	NGNO/MNC
8d. Katershoeve [ZWB-4]	S	-	-	+	-	[+]	-	NGNO/MNC
9. Zwart Water [ZWW]	C	-	-	+	-	-	-	NGNO

Discussion

Despite the alarming state of biodiversity in Belgium [e.g., Maes & Van Dyck 2001], the use of detailed species-specific knowledge and appropriate, often small-scaled management measures to ensure the survival of threatened species, is still in one's infancy in Belgium [Van Dyck *et al.* 1999]. The data collected on the butterfly's distribution and changes therein, its host plant and habitat, population sizes and on mobility and colonization capacity, allows us to define functional conservation units [FCU] to organize and prioritize the conservation of the threatened large blue butterfly *M.alcon* in Belgium. In this sense, conservation units as defined here are pragmatic tools based on scientific species-specific evidence. Although we have not verified it at the population genetic level, the FCU-approach is likely to resemble the concept of evolutionary significant units [ESU; Ruckelshaus *et al.* 2003]. An ESU is a population that is reproductively isolated from other conspecific population units, and which represents an important component in the evolutionary legacy of the species [Meffe & Carroll 1997]. Before we discuss the different FCU's and the associated conservation measure programs, we firstly interpret our results on the state of the Belgian populations of *M. alcon* and results on mobility and colonization capacity.

The critical state of the Belgian *M. alcon* populations

Although most of the former *M. alcon* populations are located in areas with a protected status, a large number of local extinctions occurred. Table 7.1 shows that the Belgian populations of *M. alcon* are actually small to very small, often located in small habitat patches, with a limited number of host plants and host ants. According to Thomas [1991], *Maculinea arion* populations with fewer than 400 adults are likely to experience periodic extinctions and populations with 400-1 000 adult butterflies can be regarded as 'safe'.

Apart from one population [Fonteintje], all Belgian *M. alcon* populations have far too small population sizes to have a reasonable perspective on a sustainable conservation [i.e., low extinction probabilities; Elmes & Thomas 1987; Hanski & Thomas 1994].

The main factors associated with the presence of *M. alcon* in Belgium are wet heathland area and the number of *G. pneumonanthe* plants [cf. Wallis de Vries 2004]. Large heathland areas have a larger habitat heterogeneity which makes them more resilient to environmental dynamics. For example, in small areas, *G. pneumonanthe* and host ant nests tend to be spatially concentrated in the lowest depressions of a site which makes them vulnerable since prolonged rainfall can drown a large proportion of the caterpillars [e.g., 176.6 mm rain in July 2000 compared to 41.4-76.1 mm in the five previous years]. Furthermore, Maes *et al*. [2003] have shown that larger wet heathlands have higher ant nest densities, which increases the necessary spatial overlap between host plants and host ant nests [Van Dyck *et al*. 2000]. The absence of a correlation between vegetation cover and the presence of *M. alcon* is probably due to the fact that populations of *M. alcon* can persist for a relatively long time after habitat degradation due to the longevity of the Marsh gentians and the time lag between changes in vegetation structure and changes in ant species composition.

Mobility, colonization and behaviour

As in most *Maculinea* spp. [Stettmer *et al*. 2001], but also in other specialised butterflies [e.g., Thomas 1985; Neve *et al*. 1996; Bergman & Landin 2002; Betzholtz 2002], a large proportion of *M. alcon* butterflies is very sedentary. Although mean distances moved did not differ between males and females, in both populations of Zwarte Beek males covered larger distances than females, contrary to Liereman. These differences can probably be explained by differences in the configuration of both sites: Liereman consists of a cluster of nearby habitat patches with many edge situations [resulting in a area/perimeter ratio of 15.8] with a prominent tree row splitting the site in two discrete flight areas [Talloen W. and Van Dyck H., unpubl. data] while Zwarte Beek populations have a more continuous habitat [with area/perimeter ratios of 31.5 and 21.3 for Fonteintje and Panoramaduinen respectively]. Host plant distribution also differs between both sites: in Liereman *G. pneumonanthe* are clustered in patches while in Zwarte Beek they are uniformly spread over the flight area. Therefore, females have to move longer distances between host plant patches in Liereman than in Zwarte Beek. This result indicates that one should be careful to interpret sexual differences in movements when based on data from one site, or even from a single year [e.g., Baguette 2003]. Host ant nest distributions were only surveyed in plots of 100 m^2 [Maes *et al.* 2003] and it may be difficult to extrapolate these densities to entire flight areas. The role of host ant nests on the female's choice of ovipositing on host plants and thus on the daily movements is still under debate [Thomas & Elmes 2001; Van Dyck *et al.* 2000, Van Dyck H. & Regniers S., unpubl. data].

MRR-studies usually underestimate dispersal distances because the chance of recapturing marked butterflies decreases with distance and the distance covered by butterflies leaving the population is usually unknown [Turchin *et al.* 1991; Shreeve 1992, 1995]. Colonization data give more relevant figures for feasible dispersal distances [cf. Baguette 2003]. The limited mobility and colonization capacity of *M. alcon* observed here are not only a species-specific trait, but also depend on the size of potential source populations and on the availability of suitable habitat patches within a certain distance of other populations [Thomas *et al.* 1998a].

The behaviour of species at the edge or even outside the habitat has become an important research topic, especially in highly fragmented landscapes [Merckx *et al.* 2003; Schtickzelle & Baguette 2003]. Behavioural responses can have important implications for the optimal design of habitat edges, stepping stones or corridors [Schultz 1998; Haddad 1999; Ricketts 2001; Ries & Debinski 2001; Schultz & Crone 2001]. For example, the Fender's blue butterfly [*Icaricia icarioides fenderi* - Schultz 1998; Schultz & Crone 2001] and the Black-veined White *Aporia crataegi* [Watanabe 1978] dispersed 2-3 times faster, and also further, outside than within suitable habitat. Recent observations in other butterfly species by Schultz [1998], Ries & Debinski [2001] and Schultz & Crone [2001] are in line with our observations in *M. alcon* of high returning probabilities of butterflies approaching the edge of their habitat: the higher the trees at the edge of the habitat, the more likely the species was to return. This knowledge can be used to manipulate the design of [or to create] physical edges to temporarily prevent individuals from leaking from a small local population [e.g., by planting tree rows around small and isolated patches], certainly when suitable habitat is unavailable within the colonization capacity [Kuussaari *et al.* 1996; Thomas *et al.* 1998b; Thomas & Hanski 1999; Betzholtz 2002]. Further experiments on behaviour at habitat boundaries and movements through the landscape matrix are required to understand the mechanisms behind particular movement patterns among different landscapes [Merckx *et al.* 2003].

Functional conservation units for *M. alcon* in Belgium

Traditional but non-specific management regimes have low chances of being beneficial for small relict populations of habitat specialists like *M. alcon*. The scale at which species-specific conservation measures are taken, has to be in accordance with the target species' ecology. We defined 'functional conservation units' [FCU] by combining data on i] detailed distribution of the butterfly, its host plant and wet heathland, ii] population sizes and iii] mobility and colonization capacity. A FCU is a spatial entity in which actual or potential habitat for the study species is available and in which specific management and restoration measures should be concentrated. In the case of *M. alcon*, we assume FCU's separated by >10 km as completely isolated [Fig. 7.4]. FCU's have to be regarded as dynamic instruments that can change both in time and in space when conditions change [e.g., absence/presence, habitat quality].

Functional Conservation Unit-1 [FCU-1]

Because 500 m was the maximum observed distance moved during our MRR study, it can be used as an upper limit for relatively frequent, daily movements within habitat. Within this range, habitat will be used almost immediately after it becomes suitable. Objectives in FCU-1 are to increase the butterfly population size by optimising actual habitat conditions [cf. Thomas *et al*. 2001], enlarging habitat patches and restoring all potential habitat. Management measures should be small-scaled and with a close attention for remaining resources. In addition to a conventional maintenance management such as low intensity grazing [1 grazer/3-10 ha - Londo 1997], small-scale burning and sod-cutting, intensive care management will be necessary in FCU-1 to increase both the densities of *G. pneumonanthe* plants and *Myrmica* ant nests [Van Dyck *et al*. 2000]. Such labour-intensive measures cannot be maintained on the long term, and should be regarded as a temporal investment to increase the number of butterflies to a safer and sustainable level. Spatial spreading and increasing densities of *G. pneumonanthe* is achieved by very small-scaled sod-cutting [m^2] and/or 'choppering' in un-grazed sites and 'combing' in grazed situations. Seeds of *G. pneu-*

monanthe are absent from seed banks and are poor dispersers [<1 m - Oostermeijer *et al*. 1992]. Therefore, sod-cutting needs to be executed in the immediate vicinity of existing *G. pneumonanthe* plants [within a radius of 20-100 cm], should not be too deep [to maintain suitable abiotic conditions for the germination of *G. pneumonanthe* seeds] and should leave the microrelief intact to enable *Myrmica* ants to rapidly colonize the sod-cut patches. However, due to atmospheric deposition, conditions at the sod-cut soil surface can be far too acid for the germination of *G. pneumonanthe* [Vanreusel and Smets 2002]. In some experimental plots, germination could, therefore, be stimulated considerably by treating the soil with lime, which is in our opinion only acceptable if it is regarded as a temporary measure.
'Choppering' [i.e., creating scattered bits of open ground by mowing into the ground with a brush cutter] imitates the trampling of cattle and creates germination sites for *G. pneumonanthe*. Finally, 'combing' [i.e., the removal of decaying litter from *Molinia caerulea* tussocks] makes young leaves of *Molinia caerulea* more accessible for grazers and therefore increases the actually grazed area by guiding grazers into formerly un-grazed patches. The newly grazed areas can become more suitable for germination, while grazing pressure will be relaxed in areas where *G. pneumonanthe* has a good chance to germinate, but only little chance to reach the flowering, adult stage due to overgrazing.
Some of the nature reserves with actual *M. alcon* populations are grazed by cattle, horses or sheep, which is an appropriate management strategy to maintain or create well-structured wet heathland. So far, managers in most reserves have only little experience in fine-tuning effects of grazing, and the pressure on particular habitat patches can be far too great for this butterfly-plant-ant system because of an underestimate of the actual grazing pressure. The exclusion of grazers between 15 July and 30 September from the *G. pneumonanthe* patches with the highest numbers of *M. alcon* eggs is an

appropriate additional intensive care measure that resulted in a threefold increase of the number of eggs in one of the populations between 2001 and 2002 [Hageven; Ghis Palmans, pers. comm.].

Functional Conservation Unit-2 [FCU-2]

The FCU-2 determines the scale at which has to be looked for potentially new habitat. Heathland patches within 2 km around occupied patches, as derived from the colonization data, have a reasonable chance to be colonised naturally when they become suitable. Within this area, habitat restoration or creation on a larger spatial scale can help develop local or regional networks of patches in a metapopulation structure [Thomas and Jones 1993]. In this respect, stepping stones seem to be better for *M. alcon*, in 'connecting' occupied habitat with other suitable patches than supposed corridors like woodland rides [Webb and Thomas 1994; Schultz 1998; own observations]. Emphasis should therefore be on restoring habitat and creating new habitat between existing populations, in order to increase network connectedness.

Functional Conservation Unit-3 [FCU-3]

The third type of functional conservation unit are networks of potential habitat in which the species is actually absent. FCU-3 sites are candidates for reintroduction programmes. These units can be divided into sites that are actually suitable [FCU-3a] and sites where the habitat can become suitable after a restoration program [FCU-3b]. All FCU-3's that meet the criteria are sites where *M. alcon* went extinct in the 1990's. Only two sites [Ziepbeek and Kalmthout] appear immediately suitable for *M. alcon* [FCU-3a: large area of wet heathland, large number of host plants, high densities of *Myrmica* ants; *M. alcon* can be considered a target species in the management schemes,

etc.]. Two other sites [Buitengoor and Maten] have a large area of wet heathland but the densities of both the host plant and *Myrmica* ant nests should be increased before considering a possible re-introduction [FCU-3b].
In both FCU-2 [where patches have a reasonable chance to be colonized in a spontaneous way] and FCU-3 [where local introductions are required], restoration management should be executed to restore presently unsuitable wet heathland patches. Since the butterfly is absent from FCU-2 and FCU-3, management measures can be executed more intensively than in actual *M. alcon* populations. Large-scale sod-cutting [100 – 1000 m^2] and a more intensive grazing regime can help to achieve a suitable starting point for wet heathland restoration. Prior to any large-scaled sod-cutting, a census on the presence of *Myrmica* ants is highly relevant. *Myrmica* ants can be present in deteriorated heathlands [Maes *et al*. 2003] and although they are relatively rapid colonizers of suitable areas, it may take a long time before a restored site provides suitable nesting and foraging habitat. Therefore, there is a considerable gain in terms of time when in inevitable large-scaled sod-cutting practice, micro-topography and some vegetation strips are spared [Brian *et al*. 1976; Mabelis 1976; Maes *et al*. 2003]. Long, relatively small strips of sod cutting and of spared vegetation are predicted to have the best potential in this respect. Additional measures in the spared vegetation stripes like particular mowing regimes can further contribute to heathland restoration without a dramatic temporal loss of local ant diversity. Further research on responses of ants to restoration measurements are required to refine these guidelines.
Re-introduction should, in our opinion, be considered as an emergency measure, but one that should be considered together with the several other strategies discussed above to deal with the precarious situation of *M. alcon* in Belgium. However, this measure has not yet been included in the regional nature conservation legislation and policy of conservation agencies. It therefore remains largely unexploited [Van Den Berge *et al*. 1995]. Scientifically underpinned re-introductions of other *Maculinea* spp. elsewhere in Europe

have shown their potential to speed up spatial risk spreading in a successful way [e.g., *M. arion* in England; Thomas 1995; and *M. teleius* and *M. nausithous* in the Netherlands; Wynhoff 1998a]. At present, the re-introduction of *M. alcon* in one of the former populations [Ziepbeek], is under investigation [Vanreusel *et al*. 2002]. In some of the present-day *M. alcon* sites, especially in large military areas such as Sonnisheide and Groot schietveld, suitable habitat patches are too far apart to have a reasonable chance of colonization on the short term. Here, translocation could be considered to spread the risks on local extinctions among an increased number of patches. It is evident that such a measure has to be accompanied by restoration measures in and among suitable patches to [re-]create a sustainable population network on the long term.

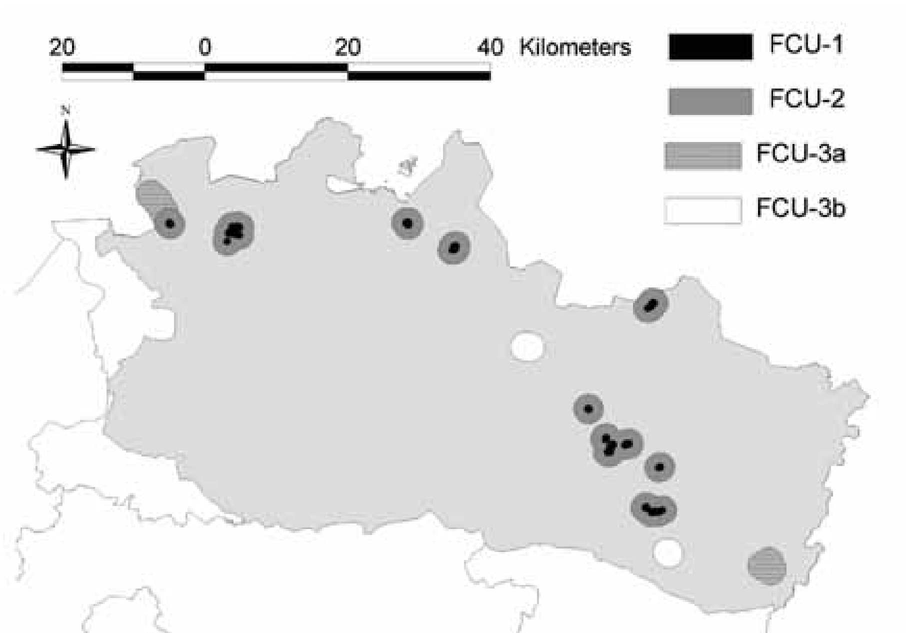

Figure 7.4. Functional conservation units for *M. alcon* in Belgium. FCU-1 = presently occupied habitat patches plus the area within a range of 500 m surrounding them; FCU-2 = the areas within a range of 2 km around the occupied habitat patches and FCU-3 = potential re-introduction sites [a = actually suitable and b = potentially suitable after restoration].

Two major gaps remain in the ecological knowledge of *M. alcon* in Belgium but also elsewhere: host ant use and genetic differences between populations. Both information sources are important to determine the best 'matching' source population for a translocation or re-introduction. It recently became clear that much more efforts are needed to study host ant use of *M. alcon* in Belgium. The Belgian populations are probably on the transition zone between *Myrmica ruginodis* and *M. scabrinodis* as optimal host ant [Elmes *et al*. 1994; Karsten Schönrogge pers. comm.]. Our own preliminary observations indicate that *M. ruginodis* is used in the majority of the populations, but other *Myrmica* ants were observed as host ant as well [*M. rubra, M. scabrinodis* and probably even *M. schencki*]. Host ant-use, genetic differentiation and patterns of pheromone profiles of caterpillars and candidate host-ants [cf. Akino *et al*. 1999; Elmes *et al*. 2002] are currently under investigation within an extended European research program.

Conclusion

The species action plan for *M. alcon* [Vanreusel *et al*. 2000] was the first action plan for an invertebrate species in Flanders [north Belgium]. This pilot project points at a more widely important issue that needs to be tackled by conservation policy: site-based conservation strategies that deny species-specific aspects are only seldomly able to preserve threatened habitat specialists. Additionally, labour-intensive and expensive species-specific measures need to be temporarily incorporated into current management schemes. The implementation of this species action plan in the field aims at both increasing the viability of the existing populations and creating new suitable sites. Although the Flemish government has invested in a species action plan for *M. alcon*, we ascertain that there is, so far, only little effort and virtually no budget to monitor and imply the proposed measures. It remains a typical and highly relevant bottleneck for conservation that policy makers are less willing to invest in constructive feed-back and implementation programs than in plans. We consider the approach of functional conservation units a useful tool to organize species-specific measures at different spatial scales in Belgium [or elsewhere] that can be similarly applied for other threatened species.

8 From single to multispecies approaches in nature conservation

foto: Jeroen Mentens

"... limited suites of species may serve as effective umbrellas for regional faunal assemblages when practitioners must prioritize areas for conservation in a managed landscape. ... we recommend that a suite of umbrella species rather than a single species be employed in conservation management. ... the contention that any species or group of species can serve as a reliable surrogate measure of diversity, management efficacy, or ecosystem integrity must be treated as a hypothesis to be rigorously tested: it cannot be assumed that proposed umbrella or indicator species do indeed signal what they are supposed to."

Erica Fleishman *et al*. **[2000].** A new method for selection of umbrella species for conservation planning. Ecological Applications 10: 569-579.

We analyzed whether a single species [i.e., the threatened butterfly *Maculinea alcon*] was a useful indicator species for the quality and quantity of wet heathlands in Belgium. We compared its indicator capacities with those of a multispecies approach in which we selected easily recognizable, intermediately rare and ecologically well-known species. During a survey of 18 wet *Erica tetralix* heathlands in Belgium, we identified 624 species from 20 different taxonomic groups. Sites with the single indicator species *M. alcon* were significantly richer in typical wet heathland species and in Red List species than sites without; the indicator species *M. alcon* however failed to indicate habitat heterogeneity [i.e., the presence of different typical wet heathland habitat characteristics]. The multispecies approach resulted in an umbrella group of nine species from five different taxonomic groups [two birds, two dragonflies, two butterflies, two plants and one grasshopper]. This umbrella group was positively correlated with the diversity of typical wet heathland species richness and with habitat heterogeneity. Furthermore, the complementary information of the umbrella group had a useful signaling function about habitat size and configuration, vulnerability to fragmentation, eutrophication, desiccation and contained species of different trophic levels; this was not the case for *M. alcon* as a single indicator species. We discuss the use of single indicator versus multispecies approaches as conservation umbrellas and advocate a much wider use of combined knowledge from different taxonomic groups in conservation planning and evaluation.

Maes D. & Van Dyck H. [submitted manuscript]. A single indicator versus a multispecies approach: a case study on wet heathlands.

Introduction

In several parts of the world, like in NW-Europe, natural landscapes became human-dominated biotopes several centuries ago [Thomas 1993]. Conservation efforts in such densely populated industrialized regions with a high pressure on both the environment and on biodiversity [e.g., Belgium - OECD 1998], are typically focused on semi-natural, traditionally managed landscape remnants or biotopes [e.g., nutrient-poor hay meadows, heathlands]. Conservation practitioners mostly rely on their experience with, and knowledge of, traditional agricultural practices in order to manage reserves [e.g., grazing, mowing or sod cutting regimes]. This approach is, however, based on the – mostly implicit – assumption that repeating a traditional management type will ensure appropriate abiotic conditions for local biodiversity persistence [Pullin & Knight 2001]. A continuous decline of several species, even in nature reserves – like for butterflies in NW Europe [Maes & Van Dyck 2001; Warren *et al.* 2001] – has stimulated the debate on the role of using species as explicit targets or as tools for the conservation of semi-natural biotopes. This contrasts with the management of ecosystems in traditional[-like] ways without reference to particular species [Simberloff 1998]. Species have the benefit that several requirements relating to habitat quality, quantity and geometry can be defined or estimated, and this ecological knowledge may consequently be used as standards for management planning and/or evaluation [Mc Geoch 1998; Hilty & Merenlender 2000]. But then the question arises, which species to work with? Available knowledge is an obvious bottleneck here. Moreover, conservation practitioners request for, preferentially, rather simple, straightforward approaches that can be readily implemented by non-experts in order to keep such efforts within their limited time and financial budgets [Fleishman *et al.* 2000]. Therefore, the use of short-cut concepts like indicator species to protect, manage or restore habitats and local biodiversity is highly attractive [Lambeck 1997]. Proposed indicator species have usually been conspicuous mammals [e.g., Wilcox 1984; Beier 1993], birds [e.g., Martikainen *et al.* 1998; Mikusinski *et al.* 2001; Rubinoff 2001] or vascular plants [e.g., Oliver *et al.* 1998; Pharo *et al.* 1999]. Some of these organisms have the additional benefit of being flagship species [i.e., attracting public and political attention more easily than others - Landres *et al.* 1988; Simberloff 1998]. However, the use of a single species or a single taxonomic group as a conservation umbrella for other sympatric species or for the integrity or quality of a certain ecosystem has been criticized [e.g., Landres *et al.* 1988; Niemi *et al.* 1997; Prendergast *et al.* 1993a], because the effectiveness of the concept has often been assumed, but is rarely tested [Andelman & Fagan 2000; Andersen 1999; Fleishman *et al.* 2001b; Simberloff 1998].

Recently, several authors have advocated a multispecies approach in conservation biology, i.e., using a group of species instead of a single indicator species [e.g., Lambeck 1997; Hilty & Merenlender 2000; Fleishman *et al.* 2001b; Root *et al.* 2003]. The underlying rationale is that a carefully selected group of species is more likely to provide a complementary, integrative picture of the quality of a reserve [or a habitat network] than a single species. Furthermore, Collins & Thomas [1991] and Samways [1993], among others, have plead for a more prominent use of insects and other invertebrates in conservation biology than is currently the case. This may particularly be appropriate in traditionally managed, man-made habitats where habitat specialist insects heavily depend on vegetation structures and associated microclimates that are not necessarily relevant to birds or mammals [Murphy & Wilcox 1986; Thomas 1994]. Hence, large species groups like insects and/or other invertebrates should not *a priori* be excluded from such multispecies approaches in order to involve ecological aspects at intermediate or even small scales [Kremen *et al.* 1993; Brown 1997; Mc Geoch 1998; Kotze & Samways 1999].

The majority of studies dealing with indicator species and with cross-taxa comparisons of species richness focus on reserve and habitat network selection, often at a rather coarse scale [e.g., van Jaarsveld *et al.* 1998; Poiani *et al.* 2000]. Depending on the type of conservation question, the spatial scale of indicator evaluation needs to be carefully considered [Pearson & Cassola 1992]. In Belgium, but also in other western countries with high pressure on the open space, decisions on the configuration of habitat networks and reserves are principally non-ecologically based [e.g., political agreements with other land-users, socio-economic priorities]. However, even in such situations, there is a growing interest in using species-specific knowledge [such as the indicator species concept] as a tool to develop and adapt habitat management and restoration plans once reserves or local habitat networks have been established [Root *et al.* 2003]. Here, we compare the indicator capacity of a single species with that of a multispecies approach as a conservation and management tool for temperate wet heathlands. In Europe, Northern Atlantic wet heaths with *Erica tetralix* are of high conservation value [EU Habitat Directive 92/43/EEC]. Wet heathlands are used as a model system for underpinning use of multispecies approach because it has been extensively studied recently. The threatened Alcon Blue butterfly [*Maculinea alcon* Denis & Schiffermüller 1775], confined to wet heathland in Belgium [Maes *et al.* in press], was tested as an indicator species of typical species richness and wet heathland quality. All *Maculinea* butterfly species are of conservation concern throughout Europe [Munguira & Martín 1999] and have a complex life history being obligate ant brood parasites. Therefore, *Maculinea* butterflies have attracted much attention in ecological and conservation biology studies [Thomas *et al.* [1998c] and references therein]. For the multispecies approach, we incorporated species from several taxonomic groups [vertebrates, invertebrates and vascular plants], and adopted selection criteria to compile a list of species covering a wide range of ecological information. Such a multispecies group with, ideally, easily recognizable and ecologically well-known species, should enable non-experts [wardens and volunteer nature managers] to evaluate the appropriateness of a reserve for habitat specialists or the success of their management measures more easily than a time consuming and extensive survey of a large number of ecologically ill-known taxonomic groups.

Methods and materials

Maculinea alcon

M. alcon has a scattered distribution in Europe [Kudrna 2002]. The Belgian distribution is limited to the Campine region in Belgium [Fig. 8.1] where it only occurs on wet heathlands with *Erica tetralix*. *M. alcon* is an obligate ant brood parasite [in Belgium mainly of *Myrmica ruginodis*; Elmes *et al.* 1994] and uses *Gentiana pneumonanthe* as a host plant. Both the butterfly and the host plant are threatened in Flanders, the northern federal state of Belgium [Biesbrouck *et al.* 2001; Maes *et al.* in press]. The selection of *M. alcon* as potential indicator species is based on its assumed indicator capacities by several authors [Bink 1992].

Sampling sites

Within the Campine region [NE Belgium – Fig. 8.1], we selected 18 wet *Erica tetralix* heathland sites: nine with a present-day population of *M. alcon* and an equal number of sites where the species was never documented [Table 8.1]. The limited number of present-day populations of the threatened *M. alcon* [twelve] determined the number of possible study sites. Two of these sites are in inaccessible military areas and one *M. alcon* population had become very small. This restricted the number of possible study sites to nine. The sites varied in size from 0.08 ha to 5.29 ha, but patches with and without *M. alcon* did not significantly differ in area [Kruskall-Wallis test H [1,18]=1.875, p>0.17]. Most common plant species on the sites were *Erica tetralix*, *Molinia caerulea*, *Calluna vulgaris*, *Gentiana pneumonanthe* and *Scirpus cespitosus*.

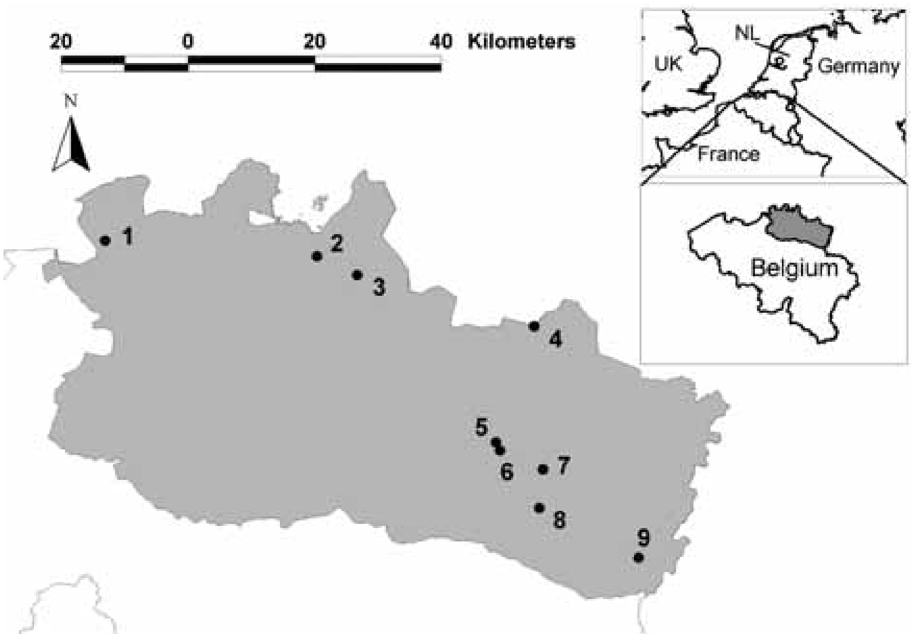

Figure 8.1. Location of the investigated sites within the Campine region [shaded in grey] in Belgium. The numbers correspond with the site numbers in Table 8.1.

Table 8.1. Investigated plots [the site numbers correspond with those on Fig. 8.1]; Site info: *M. alcon*: + = present, [+]= extinct recently [1998], but scored as present in the analysis, - = absent; PA = patch area [in ha]; SA = Site area [in ha]; Status: FNR = Flemish Nature Reserve, PNR = Private Nature reserve, PP = Private property, MA = Military area; Species info: number of species found in the different plots - All = all species; RL = Red List species; TWH = Typical wet heathland species. Dist. : distance between patches in the same site in meters.

		Site info			Species info			
Site [Locality]	*M. alcon*	PA	SA	Status	All	RL	TWH	Dist.
1. Kalmthoutse heide [Kalmthout]								3082
1.a - WIT-1	+	2.59	72	FNR	174	18	25	
1.b - KAL-4	-	1.32	836	FNR	147	13	25	
2. Zwart water [Turnhout]								1359
2.a - ZWW-1	+	2.69	21	PNR	239	18	25	
2.b - KOE-1	-	1.07	8	PP	170	13	21	
3. Liereman [Oud-Turnhout]								1073
3.a - LIE-2	+	1.75	175	PNR	162	25	28	
3.b - LIE-3	-	4.34	175	PNR	133	13	19	
4. Hageven [Neerpelt]								426
4.a - HAG-5	+	0,68	205	PNR	154	23	28	
4.b - HAG-15	-	0.08	205	PNR	175	20	15	
5. Panoramaduinen [Hechtel-Eksel]								401
5.a - ZWB-3	+	2.99	2746	MA	143	21	26	
5.b - ZWB-6	-	1.94	2746	MA	205	25	21	
6. Fonteintje [Koersel-Beringen]								403
6.a - ZWB-2	+	5.29	2746	MA	175	23	33	
6.b - ZWB-5	-	0.99	2746	MA	172	20	29	
7. Sonnisheide [Houthalen-Helchteren]								230
7.a - HHH-1	+	1.24	2183	MA	185	26	26	
7.b - HHH-3	-	2.02	2183	MA	250	21	24	

		Site info			Species info			
Site [Locality]	*M. alcon*	PA	SA	Status	All	RL	TWH	Dist.
8. Teut [Zonhoven]								949
8.a - TEU-1	+	4.77	317	FNR	164	23	30	
8.b - TEU-3	-	0.40	317	FNR	147	12	23	
9. Vallei van de Ziepbeek [Zutendaal]								1094
9.a - ZIE-3	[+]	1.06	170	FNR	141	24	27	
9.b - ZIE-1	-	2.02	170	FNR	144	24	24	

Sampling methods

Appropriate techniques were used to sample 20 faunal groups [Table 8.2]. In each site, we placed three pitfall traps [Southwood 1978] and three white water traps at a 10 m interval on 30 March 2000; water traps were emptied fortnightly until 30 September 2000, pitfall traps until 15 March 2001. Sweep net samples were taken over a length of 15 m and fixed transects were walked over a length of 50 m parallel to the pitfall and water traps twice a month from May to August 2000 [Pollard & Yates 1993]. Additionally, we visually searched for easily recognizable species during 30 min twice a month from May to August 2000. We applied threshold values [i.e., minimum numbers observed] for several species [in particular for invertebrates] to establish the presence of a local population [e.g., > 5 individuals for butterflies, grasshoppers, etc.]. Finally, in each site, vegetation surveys were made during the summer of 1999 and 2000 in four plots of 2 m x 2 m. Species caught by pitfalls, water traps and sweep net were sorted out in the laboratory and classified with a binocular microscope; species seen on transect walks, during visual searching and in the vegetation surveys were identified in the field. In all sites, we measured habitat patch area

using a GPS [precision 1 m] and we scored the absence or presence of seven habitat characteristics of wet heathland: soil humidity [permanently wet = 1, dry in summer = 0], bare ground [important for germination of typical plants and ground dwelling invertebrates: present = 1, absent = 0], scattered trees [important for insectivorous birds and territories of butterflies: present = 1, absent = 0], moorland pools [important for aquatic invertebrates of nutrient poor water, e.g., dragonflies; dolichopodid flies: present = 1, absent = 0], microtopography [important for variation in microclimatic conditions: present = 1, absent = 0], seepage [important for the compensation of nutrient rich deposition: present = 1, absent = 0] and particular *Sphagnum* mosses [indication of undisturbed wet heathland [Schaminée *et al*. 1995]: present = 1, absent = 0]. Habitat heterogeneity is subsequently expressed as the sum of the different habitat characteristics.
Red Lists are available for 11 of the investigated taxonomic groups in Flanders [Table 8.2]. A Red List species is a species that belongs to the Red List categories 'Extinct', 'Critically endangered', 'Endangered' or 'Vulnerable'. For our purpose, a typical wet heathland species is a species that is confined to, or has its highest densities in, wet heathland in Belgium. The full list of 68 references used to identify the species, assess the Red List status and classify species as typical for wet heathland can be obtained from ftp://ftp.instnat.be/Users/Dirk_M/wetheathlandindicator.rtf.

Table 8.2. Sampling method [O = Observations, P = Pitfalls, S = Sweep net, T = Transect counts, V = vegetation surveys, W = water traps] and number of species [All], Red List species [RL: - = no Red List species present, . = no Red List available] and typical wet heathland species [TWH] for the different taxonomic groups.

Taxonomic group	Sampling Method	All	RL	TWH
Amphibians and reptiles	O/[P]	8	-	2
Ants [*Hymenoptera - Formicidae*]	P/W	27	.	1
Birds	O/T	25	8	4
Bugs [*Hemiptera – Heteroptera*]	P/W/S	38	.	-
Burying beetles [*Coleoptera – Silphidae*]	P/W	4	.	-
Butterflies [*Lepidoptera – Rhopalocera*]	O/T	24	6	3
Carabid beetles [*Coleoptera – Carabidae*]	P/W	71	7	9
Centipedes [*Myriapoda*]	P/W	5	.	-
Cockroaches [*Dictyoptera – Blattodea*]	P/W	4	.	-
Day flying moths [*Lepidoptera* partim]	O/S/T	13	.	1
Dolichopodid flies [*Diptera – Dolichopodidae*]	W/P/S	25	1	4
Dragonflies [*Odonata*]	O	37	12	12
Empidid flies [*Diptera – Empididae*]	W/P/S	19	2	1
Grasshoppers [*Orthoptera*]	O/S/P/T	16	3	2
Hoverflies [*Diptera – Syrphidae*]	W/P/S	41	.	4
Leafhoppers [*Hemiptera – Homoptera*]	P/W/S	34	.	-
Mammals	P	8	-	-
Vascular plants	V	33	3	13
Sphecid wasps [*Hymenoptera – Sphecidae*]	W/P/S	4	.	-
Spiders [*Araneae*]	P/W/S	188	56	8
Total		624	98	64

A multispecies approach for wet heathlands in Flanders

Hilty & Merenlender [2000] outlined a step-wise decision-making framework for the compilation of a set of taxonomically diverse indicator species. We slightly modified this concept incorporating recommendations in Landres *et al.* [1988], Mc Geoch [1998], Caro & O'Doherty [1999], Fleishman *et al.* [2000], Poiani *et al.* [2000] and Fleishman *et al.* [2001b]:

Step 1 - Decide what ecosystem attributes indicator taxa should reflect

A wet heathland with high conservation value can be defined as being large and containing necessary habitat characteristics for a variety of habitat specialists. Therefore, a multispecies group should contain species that need large areas of wet heathland [to Belgian standards], that are sensitive to fragmentation, desiccation and eutrophication, and that are dependent on one or more of the typical habitat characteristics as stated above. As a whole, the multispecies group should encompass all of the habitat characteristics more than once, but it is not necessary that every single species in the multispecies group does so.

Step 2 - List all species or taxonomic groups that meet baseline information criteria

Baseline information was considered sufficient when taxonomy is clear, biology and life history are well studied, the species' distribution is sufficiently well known, the tolerance levels to environmental pressures are known and the correlation to ecosystem changes is established.

Step 3 - Use only intermediately rare and easily detectable species, that are evenly distributed in the focal area

After Step 2, we only retained easily observable species [during the day, no trapping devices needed] and identifiable by non-experts [using a field guide and or binoculars], that are intermediately rare [Fleishman *et al.* 2000], i.e., between 20-60 mapping grid cells [5 x 5 km] [i.e., 10-30% of all grid cells in the Campine region] and that are homogeneously distributed in the focal region.

Step 4 – List available information on niche and life history and on sensitivity to environmental stressors

Niche and life history criteria concern trophic level, reaction time to environmental changes, mobility, minimum area requirements, detailed niche of the species [necessary structural habitat characteristics] and the sensitivity to different environmental stressors [eutrophication, desiccation, fragmentation, etc.].

Step 5 – Compile a set of complementary species from different taxonomic groups to satisfy every criterion from Step 1 by more than one taxon

From the list obtained after Step 4, a group of species was selected that is complementary [all criteria of Step 1 should preferably be present at least twice] and that consists of species of different taxonomic groups.

Analysis

Differences in overall species richness, Red List species richness and typical wet heathland species richness among sites with and without *M. alcon* were tested with a paired t-test for dependent samples [Sokal & Rohlf 1995]. *M. alcon* itself was excluded from the number of species on sites where the butterfly was present. Data were $\log_{10}$-transformed prior to analysis to obtain normality. We tested whether the multispecies group was correlated with the number of typical habitat characteristics and with species diversity [all, Red List or typical wet heathland species] by means of a one-tailed Spearman Rank correlation [Sokal & Rohlf 1995]. All analyses were done with the STATISTICA 6.0 software package [StatSoft Inc. 2001].

Results

Species numbers differed considerably between sites, varying from 133-250 for overall species richness, from 12-26 for Red List species and from 15-33 for typical wet heathland species [Table 8.1]. Spiders were the most species-rich taxonomic group while burying beetles, cockroaches and sphecid wasps were only represented by four species. Plants and dragonflies were well represented in the typical wet heathland species [Table 8.2].

The patch size of the wet heathlands we studied was not a surrogate for the total number of species [Spearman $r=0.041$, $p>0.87$], the number of Red List species [Spearman $r=0.083$, $p>0.74$], the number of typical wet heathland species [Spearman $r=0.224$, $p>0.37$] or habitat heterogeneity [Spearman $r=0.198$, $p>0.43$]. Habitat heterogeneity rather than patch area was correlated with the number of species, indicating that habitat quality is at least as important as habitat size in the fragmented wet heathlands in Belgium [Thomas *et al.* 2001].

M. alcon as single indicator species

Sites with *M. alcon* were significantly richer in Red List species and in typical wet heathland species but not in overall species diversity. Habitat heterogeneity did not differ significantly between sites with and without *M. alcon* [Table 8.3a].

Patches with and without *M. alcon* did not differ significantly in vegetation cover of the most abundant plant species [*Erica tetralix*, *Calluna vulgaris*, *Molinia caerulea* and *Scirpus cespitosus*; t-test, $p \geq 0.31$], except for *Gentiana pneumonanthe* that was more abundant on sites with *M. alcon* [t-test, $t=2.828$, $p=0.03$]. Sites with and without *M. alcon* did not differ in isolation [distance to nearest wet heathland site] and the differences in species numbers between sites with and without *M. alcon* could therefore be attributed to habitat quality differences [cf. Thomas *et al.* 2001].

Table 8.3. a] Average number of all species, Red List species, typical wet heathland species and habitat characteristics in sites with and without *M. alcon* and the results of the t-test for dependent samples. b] One tailed Spearman Rank correlations between the number of species from the multispecies group and the overall number of species, the number of Red List species, the number of typical wet heathland species and habitat heterogeneity [i.e., the number of habitat characteristics, n=7].

a] *Maculinea alcon*	Present	Absent	t-test	p-value
All species	170.0 ± 29.4	171.4 ± 36.6	-0.026	0.980
Red-List species	21.6 ± 2.7	17.9 ± 5.2	2.389	0.044
Typical wet heathland species	26.8 ± 2.5	22.4 ± 3.7	3.406	0.009
Habitat heterogeneity	4.0 ± 1.7	3.1 ± 1.4	1.042	0.328
b] Multispecies group	**Spearman r**	**p-value**		
All species	0.326	0.093		
Red-List species	0.209	0.203		
Typical wet heathland species	0.442	0.033		
Habitat heterogeneity	0.445	0.032		

Table 8.4. Remaining species after Step 3 of the multispecies approach for wet heathlands in Belgium and information on area and structure requirements, life history criteria and vulnerability for environmental stressors. Reaction time is expressed as a function of the number of offspring per year: Slow [one generation per year], Intermediate [one generation per year but relatively high number of eggs or young], Fast [more than one generation per year or high numbers of eggs or offspring per year]; Str.C. = Structure characteristics: Su = different succession stages; T = scattered trees; F = fens, Se = seepage; R = microtopography; W = permanently wet; Sp = Sphagnum mosses; Pr. = Pressure [sensitivity to environmental pressure]: F = fragmentation, D = desiccation, E = eutrophication, M- = sensitive to intensive management, M+ = reacts quickly to management measures. Species marked with an asterix are part of the multispecies group for wet heathland.

	Trophic level	Reaction Time	Mobility	Area	Str.C.	Pr.
Amphibians and Reptiles						
Lacerta vivipara	Insectivore	Intermediate	Low	<5ha	Su	F
Birds						
Anthus trivialis	Insectivore	Slow	High	5-25 ha	T	F
Lullula arborea	Insectivore	Slow	High	5-25 ha	T	F
*Numenius arquata**	Insectivore	Slow	High	>25 ha	W	F
*Saxicola torquata**	Insectivore	Slow	High	5-25 ha	T	F
Butterflies						
*Callophrys rubi**	Herbivore	Fast	Low	<5 ha	Su/T	D/E/F
*Plebeius argus**	Herbivore	Fast	Low	<5 ha	Su	D/E/F
Dragonflies						
*Ceriagrion tenellum**	Insectivore	Fast	Low	<5 ha	F/Se/R	D/E/F
*Leucorrhinia dubia**	Insectivore	Fast	Low	<5 ha	F	D/E/F
Grasshoppers						
*Metrioptera brachyptera**	Insectivore/Herbivore	Fast	Low	<5 ha	Su	F/M$^-$
Plants						
*Narthecium ossifragum**	.	Slow	.	<5 ha	Sp/Se/W	D/E
*Rhynchospora alba/fusca**	.	Fast	.	<5 ha	Su/W	D/E/M$^+$

A multispecies approach for wet heathland in Flanders

Ninety species were considered as typical for wet heathland in Belgium. We found 64 of these species during our survey and we considered the baseline information as sufficient for 52 of these species [Step 2]. Applying Step 3 to the 52 remaining species [easily detectable and classifiable species of intermediate rarity], only left 14 species as candidates for a multispecies group. For these 14 species, the best available information on niche, life history and sensitivity to environmental pressures was gathered [Step 4, Table 8.4]. From this list we selected a group of species in which all selection criteria from Step 1 were met with by more than one taxon. If species carried the same information, the most conspicuous and easiest to classify or observe was chosen. Finally, we selected nine species as the multispecies group for wet heathland in Flanders: two birds [*Numenius arquata* and *Saxicola torquata*], two butterflies [*Callophrys rubi* and *Plebeius argus*], two plant species [*Narthecium ossifragum* and *Rhynchospora* spec.], two dragonflies [*Ceriagrion tenellum* and *Leucorrhinia dubia*] and one grasshopper [*Metrioptera brachyptera*].

The statistically best multispecies group would have consisted of four species that individually correlated significantly with a high number of other typical wet heathland species [Mann-Whitney U-test $p<0.05$]: the Linyphiid spider *Araeoncus crassiceps*, the plants *Eriophorum angustifolium* and *Narthecium ossifragum* and *M. alcon*. However, three of these species do not meet the criteria outlined in Step 3 to be retained as suitable species for a multispecies group [the spider *Araeoncus crassiceps* is difficult to identify by non-experts, *Eriophorum angustifolium* is too common and *M. alcon* is too rare].

The multispecies conservation umbrella

The number of all typical wet heathland species [n=64] was positively correlated with the number of typical habitat characteristics, i.e., sites with greater habitat heterogeneity had higher numbers of typical wet heathland species [Fig. 8.2]. The subset of species of the multispecies group in the different sites remained positively correlated with the number of typical wet heathland species and with habitat heterogeneity, but not with the overall number of species or the number of Red List species [Table 8.3b]. Sites with *M. alcon* tend to have a larger number of species of the multispecies umbrella group [Table 8.3a].

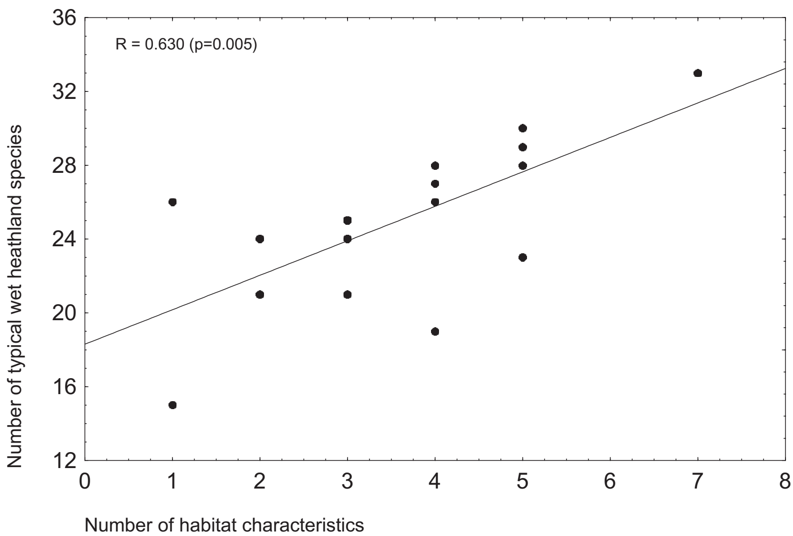

Figure 8.2. Correlation between the number of habitat characteristics [x-axis] and the number of typical wet heathland species [y-axis].

Discussion

Nature conservation in Belgium is largely biotope-based [i.e., sites are acquired and management for the presence of certain biotope types; e.g., EU Habitat Directive] and/or ecosystem-based [i.e., sites are managed in function of ecological processes; e.g., nutrient cycles, hydrology]. Several authors have shown that species can go extinct under such site- or ecosystem-based conservation policies [Pickett *et al.* 1992; Simberloff 1998]. The incorporation of species into decisions about site selection or management measures is, up to date, rather scarce in Belgium [Van Dyck *et al.* 1999]. The integration of species information and site conservation can render nature conservation considerably more efficient through the use of species as tools for site selection, management evaluation and/or the evaluation of nature conservancy policy [Lawton 1997]. Species can, because of their [international] threat status, their functional role in ecosystems, etc. also act as goals themselves [e.g. species action plans; Simberloff 1998]. This typically regards locally or more widely threatened or rare species.

The implementation of evidence-based conservation biology on conservation practice still appears to be relatively limited compared to the more widespread use of experience- and tradition-based management. Several authors have recently referred to this problem as the gap between conservation science and practice [Deem *et al.* 2001; Pullin & Knight 2001; Robertson & Hull 2001]. There is a considerable risk that conservation practitioners will consider management techniques as a target in their own and not as a tool to improve or maintain conditions for local biodiversity. Particularly for the management of semi-natural, traditionally managed biotopes [like heathlands in Europe], there is a growing awakening of the need to take species-specific requirements into account. These requirements are not necessarily guaranteed by simply restoring traditional management techniques or by maintaining ecological and abiotic processes in threatened biotopes [Pullin & Knight 2001]. In fact, traditional agricultural techniques were never intended or developed to enlarge or to conserve biodiversity. Moreover, their present-day impact most probably differs from that in the past because the environmental quality [nitrogen deposition] and the landscape context [fragmentation] have changed considerably [Thomas *et al*. 1998a; Van Dyck & Matthysen 1999]. Therefore, it is highly relevant to draw species-specific requirements with respect to habitat quality, quantity and geometry into the focus of management and conservation policy [Niemi *et al*. 1997].

Although attractive as a concept, the short-cut of a single indicator species as a surrogate for the diversity of other species or planning conservation measures for entire biotopes, has been called into question by several authors [e.g., Landres *et al*. 1988; Andelman & Fagan 2000]. Our results suggest that both the threatened *M. alcon* and the multispecies umbrella have capacities as indicators for typical wet heathland species diversity. But, only the multispecies group was an adequate indicator of habitat heterogeneity, whereas *M. alcon* alone failed to do so. Furthermore, we believe that the multispecies approach applied here meets the suggestions made by several authors to bring science closer to conservation practitioners [Deem *et al*. 2001; Pullin & Knight 2001; Robertson & Hull 2001]: the species of the multispecies group are easily recognizable by non-experts and at the same time provide information on other threatened or typical species and on habitat quality [expressed as the number of typical habitat characteristics]. Additionally, the information content of the multispecies group can be explicitly used in the evaluation or the set up of conservation actions [Mc Geoch 1998]. The presence or absence of specific species of the multispecies group should be used as an early warning function: the absence of both species that need relatively large wet heathlands, for example, indicates an insufficient continuous habitat patch; conservation practitioners can subsequently use this information to imply adequate management measures to enlarge or connect existing habitat patches. However, even the use of a group of taxonomically different species for the planning or evaluation of conservation measures remains a simplification following from inevitable pragmatism for conservation practice. But these multispecies groups have the clear benefit of forcing managers 'to cross' taxonomic boundaries and hence to explicitly take different requirements [including different scales] that are relevant for different biodiversity components into account. As the composition of multispecies groups strongly relies upon available knowledge on taxonomy, distribution and ecology, the use of multispecies approaches represents a continuous process rather than a one-off operation [Fleishman *et al*. 2001b]; additionally, one or a few species [that are absent in some sites] can locally be interchanged by other species with the same 'information content'. Other authors have proposed statistical ways to select indicator [e.g., the umbrella index; Fleishman *et al*. 2000] but this index is only applicable within taxonomic groups and cannot be used to determine umbrella species from a taxonomically diverse dataset [Fleishman *et al*. 2001b].

The multispecies approach applied here can be used in several conservation applications, e.g., evaluation of habitat quality, impact of nature management, setting conservation priorities, etc. Non-experts can evaluate habitat heterogeneity using the multispecies umbrella group on a large scale and in a relative short time period since all species are easily recognizable and detectable. The impact of nature management on the species composition of wet heathland can be evaluated by monitoring not only presence/absence of the multispecies group, but by incorporating abundances of the different species; increasing abundances of species that indicate a divergence from the presupposed goal can be used to alter the actual nature management scheme. A further extension of the multispecies approach is that it allows the prioritization of sites in a focal region: counting the number of species from the multispecies group per km^2 [the smallest grid unit used in mapping schemes in Belgium] rapidly indicates the most important wet heathlands [Fig. 8.3]; interpreting the absence of certain species from the multispecies group in intermediately [4-6 species of the multispecies group present] or low quality [1-3 species of the multispecies group present] rated wet heathlands, can indicate appropriate management measures or acquisition policies for surrounding sites to fulfill the needs of the missing habitat specialists.

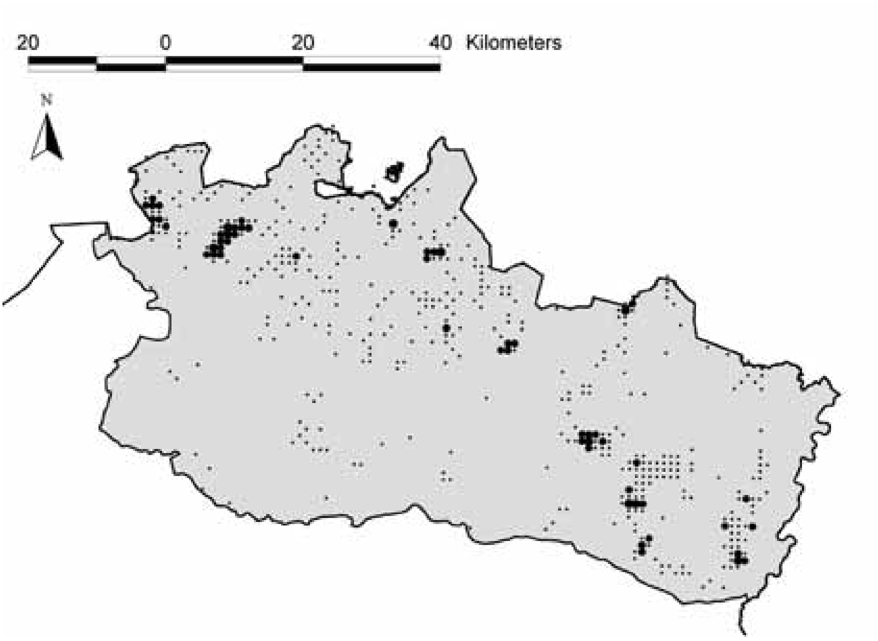

Figure 8.3. Number of species from the wet heathland multispecies group per km^2 in the Campine region [shown in grey] in NE Belgium. Small dots = 1-3 species; intermediate dots = 4-6 species; large dots = 7-9 species.

Conclusion

The use of the combined information of a well selected, but limited set of species is a practical tool for conservation practitioners and/or policy makers. Since multispecies approaches explicitly include different aspects of biodiversity [different taxa, scales, habitat demands, etc.] they can, additionally, serve as educational tools to widen managers' views on biodiversity. The multispecies group for wet heathland presented here, has the potential of being a useful pragmatic guideline with a signaling function for environmental stressors on the one hand and for different habitat characteristics on the other. Explaining the ecological mechanism underlying the signals coming from the multispecies group can subsequently be investigated through more in-depth scientific research. Such interactions between field practitioners and scientists can considerably reduce the distance between both important actors in conservation biology [Pullin & Knight 2001].
The single species we tested [*M. alcon*] had some capacity as an indicator for the richness of other typical wet heathland species, but not for habitat heterogeneity as it was tested here; additional disadvantages of *M. alcon* were its rarity and therefore its limited geographical applicability as indicator species. On the other hand, the multispecies group with nine species of five different taxonomic groups was an indicator both for other typical wet heathland species and for habitat heterogeneity; the use of only easily recognizable species that are not extremely rare made this multispecies group widely applicable and practicable for non-experts. We encourage testing the multispecies approach presented here in other biotope types and for different nature conservation purposes on two conditions: i] the ecology and the distribution of potential indicator species should be well-known and ii] the initial aims - what should the multispecies umbrella indicate – should be clearly stated [Hilty & Merenlender 2000].

9 General discussion

FOTO: Yves Adams

"... any decision about what state to manage an ecosystem for in conservation is arbitrary. The best we can do is to try and minimise modern human impacts that impinge upon the system from without, and to keep Nature's options open. Management in small reserves [from a few 10's to a few 1000 ha] is often dominated by the need to maintain habitats for one or a handful of endangered species, and more resembles gardening than anything else. And of course, deciding which species to nurture has more to do with species charisma, and human preferences than science."

John Lawton [1997]. The science and non-science of conservation biology. *Oikos* 97: 3-5.

Flanders has a limited area of conservation zones, including military areas [25,645 ha or 1.9% of the Flemish territory]. The average size of a conservation area in Flanders is 31.7 ha and only 7% of the areas is larger than 100 ha [Decleer & Vanroose 2003]. In Flanders, conservation areas are usually designated on the basis of non-ecological arguments [political agreements with other land users [e.g., military areas], socio-economic priorities [e.g., farmers], etc.]. Even in such situations, there is a growing interest in using species-specific knowledge as a tool to develop and adapt habitat management and restoration plans once reserves or local habitat networks have been established [Coppolillo *et al.* 2004; Root *et al.* 2003]. Presently, nature conservation in Flanders is largely biotope-based [i.e., sites are acquired and management for the presence of certain biotope types; e.g., EU Habitat Directive] and/or ecosystem-based [i.e., sites are managed in function of ecological processes; e.g., nutrient cycles, hydrology]. Several authors have shown that species can go extinct under such site- or ecosystem-based conservation policies [Pickett *et al.* 1992; Simberloff 1998]. The incorporation of species into decisions about site selection or management measures is, up to date, rather scarce in Flanders [Van Dyck *et al.* 1999]. The necessity and complementary nature of both species and ecosystem conservation simultaneously, however, is adequately expressed by Lawton [1997]: "...*we must therefore do what we can now to preserve both species and ecosystems; ecosystems because species need them in the short-term, and species because they make ecosystems in the long term*". The integration of species information and site conservation can indeed render nature conservation considerably more efficient through the use of species as tools for site selection, management evaluation and/or the evaluation of nature conservancy policy. Species can, because of their [international] threat status, their functional role in ecosystems, etc. also act as goals themselves [e.g., species action plans; Simberloff 1998]. This typically regards locally or more widely threatened or rare species.

In the rare cases where species are used as tools [e.g., the European obligation to delimit Ramsar and/or Bird Directive areas], there is a strong bias towards vertebrates [especially birds]. Since the assumption that a single taxonomic group conserves other species simultaneously has proven to be wrong [e.g., Prendergast *et al.* 1993a; van Jaarsveld *et al.* 1998] we, therefore, have to stress the use of a wider and more complementarity set of species in nature conservation.

Species information [e.g., distribution, ecology, threat status, etc.] is vital in nature conservation [Pullin 2002b; Simberloff 1998]. Yet, this information, although available for a growing number of taxonomic groups, largely remains unapplied in planning and evaluating conservation activities [Cort 1996; Prendergast *et al.* 1999]. The increasing speed at which species are declining or even go extinct, also in Flanders [e.g. **Chapter 3**], caused an increasing demand from practitioners for more scientifically underpinned nature conservancy policies and practical conservation actions [Pullin 2002a]. Two major problems arise applying such evidence-based approaches: 1] there are a large number of threatened species which implies that selections have to be made and 2] conservation actions are usually very urgent and do not allow long-term scientific research [Pullin & Knight 2001; Salafsky *et al.* 2002]. The previous chapters presented methodologies and case studies on how species information [both on the regional level and on the biotope level] can contribute to a more evidence-based nature conservation. In this final chapter, we discuss the added value of explicitly using species information in both policy making and in conservation practice and we will propose scientific methods and standards for the use of species in nature conservation in Flanders.

Here, we focus on the complementary role of invertebrates in particular. Four major traits are assets of invertebrates in applied conservation practices: they occupy narrow niches [habitats] within their biotope, they can persist on very small habitat patches that remain suitable for short time periods, many species are much more sedentary than birds or mammals and they usually have to complete their life cycle every year [Mc Geoch 1998; New 1995c; Thomas 1994].

Three major avenues will be treated in this general discussion:

- A more efficient use of available species information in nature conservation and delimiting priorities for gathering new information in function of nature conservation needs;
- Communication about and education on the use of species information in nature conservation.
- Towards evidence-based nature conservation in Flanders

Available species information and the need for new information

Mapping schemes

At first sight, a reasonable amount of basic species distribution information is available in Flanders [Table 1.1]. Different taxonomic groups have mapping schemes and the information is usually digitally stored. But, although some of the distribution atlases used a large number of records, the coarse grain nature [grid cell size of 5x5 km for the UTM-projection or 4x4 km for the IFBL-projection] gives a false impression of extensive coverage of the focal region [Cowley *et al.* 1999]. Although this information is present, but in some cases difficult to access for nature conservation purposes, it is rarely explicitly used in decision making because a formal protocol to do so is lacking [e.g., site selection for the Flemish Ecological Network]. Well-known exceptions of explicit [vertebrate] species use in delimiting conservation areas are the Ramsar sites for waterbirds and overwintering geese where sites where at least 1% of the global population overwinters, have to be

delimited by the subscribers of the Ramsar Convention [Devos *et al.* 1999].
A closer look at distributional information shows that it is strongly fragmented, both spatially, temporally and taxonomically [Van Dyck *et al.* 1999]. Distribution atlases and Red Lists exist [or are in preparation] for all vertebrate taxa and vascular plants. The large group of invertebrate taxa are - and always will be - strongly under-represented due to the large numbers of species, the limited number of classification keys and the necessity of special equipment for capturing and classifying invertebrates [e.g., pitfalls, microscopes, etc.]. This makes extensive mapping schemes for many invertebrate groups inaccessible to a large number of naturalists.
Spatial fragmentation of information is caused by the fact that the different mapping schemes are dispersed over different instances without a 'coordinating umbrella'. Distribution data in Flanders are collected by scientific institutes [Royal Belgian Institute of Natural Sciences [KBIN], Institute for Forestry and Game Management [IBW], Institute of Nature Conservation [IN], ...], volunteer working groups [Flo.Wer [vascular plants], Gomphus [dragonflies], SALTABEL [grasshoppers], ARABEL [spiders], FORMIDABEL [[ants], ...] and/or non-governmental organisations [Natuurpunt [mammals]]. Many of the mapping schemes originate from volunteer projects in which particular taxonomic groups were put forward because of preferences of the person co-ordinating the mapping project [supply-led] and not necessarily because of an explicit need for such data in nature conservation [needs-led]. The data collected in most of these mapping schemes are primarily used for making rather coarse scaled distribution atlases [e.g., Bauwens & Claus 1996; Decleer *et al.* 2000; Dekoninck *et al.* 2003; Maes & Van Dyck 1999; Verkem *et al.* 2004].
Temporal differences between mapping schemes are caused by the fact that the different co-ordinating bodies all have different time schedules and objectives on the one hand, but also by the fact that different taxonomic

groups require different time periods to complete their mapping schemes [e.g., plants vs. birds].

The development of centralised data bases, both on a national and on an international level, with all available distribution information would render the application of distribution data in nature conservation much more accessible for possible end users [cf. the Natuurloket in The Netherlands, http://www.natuurloket.nl]. In a first phase, the taxonomic groups for which information is already stored digitally and that have a sufficient geographical coverage can form the basis for this data base in Flanders. Subsequently, other taxonomic groups can be added depending on the speed at which they progress in gathering distribution data. This approach will limit the initial taxonomic groups to vascular plants, dragonflies, grasshoppers, butterflies, fish, amphibians and reptiles, birds and mammals. In a next phase, other groups like ants, spiders, carabid beetles, ladybirds and hoverflies could be added. As can be seen, the large group of invertebrates is, in the first phase, strongly under-represented. This situation is unlikely to change in the near future without investments in additional invertebrate conservation biologists and without the publication of standard books [both on classification and on ecology] on these invertebrate groups that may stimulate more volunteers to survey them. In order to cover a wider scope of taxonomically and ecologically different organisms in nature conservation applications [multi-species approaches, see further], such investments are urgently needed.

Many studies dealing with indicator species or with cross-taxa comparisons of species richness focus on reserve and habitat network selection, often at a rather coarse scale [e.g., Poiani *et al.* 2000; van Jaarsveld *et al.* 1998]. Therefore, the spatial scale of indicator species use evaluation needs to be carefully considered [Fleishman *et al.* 2003a; Pearson & Cassola 1992]. On the scale of Flanders, distribution data could be used for delineating key-regions in ecological networks. A possible approach to locate key-regions is the so-called hotspots approach [Myers *et al.* 2000]. In **Chapters 3-5**, hotspots are determined as the sites with the highest species richness in Flanders, but other criteria for determining hotspots can be applied as well [local species specificity, endemism, richness of threatened species, ...; Balmford 1998; Curnutt *et al.* 1994; Williams *et al.* 1996]. However, a higher resolution of the distribution information [depending on the focal species] would greatly improve the utility of distribution data [Cabeza & Moilanen 2001]. The need of using such detailed distribution data can be given by the Grizzled Skipper *Pyrgus malvae*, a threatened butterfly species in Flanders [with only two populations left]. Fig. 9.1a gives the distribution of this species on a km^2 scale [which is smaller than the one usually given in distribution atlases]. On this map, its distribution seems to coincide with the designated areas for the Flemish Ecological Network [first phase]. A closer look, however, shows that on the parcel level, all patches with populations of this threatened species fall outside the Flemish Ecological Network [Fig. 9.1b]. This example clearly shows that the delimitation of conservation areas can benefit greatly from the use of more detailed distribution data.

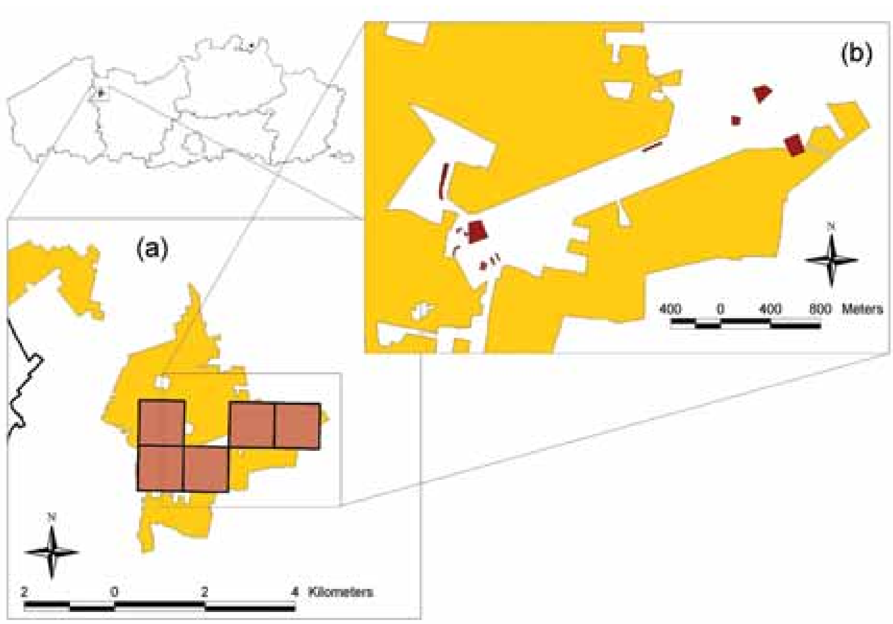

Figure 9.1. Distribution of the Grizzled Skipper *Pyrgus malvae* [in brown] on a km^2 scale [a] and on the parcel scale [b] in Drongengoedbos; in orange the Flemish Ecological Network.

On the scale of Flanders, modelling techniques could be applied to maximise the use of distribution data in nature conservation [e.g., predict potentially species-rich regions in Flanders]. **Chapters 4-5** have shown that species richness modelling can be a fairly reliable technique to incorporate un-surveyed mapping grids into nature conservancy policy making. A minimum number of well surveyed mapping units makes it possible to produce species richness patterns for a larger region, but can also be applied to predict possible distributions of individual species. Further research will have to determine the minimum number of squares and their spatial configuration necessary for a reliable modelling of species and diversity distribution. To be reliable for nature conservation purposes, a testing phase, preferably on an independent evaluation data set, is indispensable in predictive modelling [Mac Nally 2000]. Furthermore, predictive models are almost never

fully able to fit all the interactions between species richness and the environment on the coarse grain scale used in most mapping schemes [grid cells of 5 x 5 km or 4 x 4 km]. This can be caused by the high degree of fragmentation of the Flemish landscape [EEA 2002a] or to variables not accounted for in the applied model [Pape Moller & Jennions 2002]. The purpose for and the scale on which such models are used in nature conservation, should therefore always be kept in mind.
A more efficient and more uniform organisation of mapping schemes could increase the use of species information in nature conservation applications considerably, both on the national and on the international level [e.g., Mapping European Butterflies; Kudrna 2002]. The Dutch breeding bird mapping scheme [SOVON 2002] or the recently finished Flemish breeding bird mapping scheme [Vermeersch *et al.* 2000] can serve as examples for mapping schemes that aim at a full coverage of a certain region: apart from gathering distribution data on a fairly coarse scale for large-scale mapping purposes, detailed species censusing in a selection of smaller grid squares yields additional information on population densities within the different mapping squares. Furthermore, all threatened species ['Critically endangered', 'Endangered' or Vulnerable' on the Red List] can be mapped on a scale that is more relevant for practical conservation purposes in order to incorporate this information in the designation or management of conservation areas. Other extensive mapping schemes should be encouraged to apply a similar procedure although specific adaptations for certain taxa [e.g., invertebrates] will be needed such as detailed censusing on a smaller scale and in fewer grid cells [Dennis & Hardy 1999; Dennis *et al.* 1999; Pollard & Yates 1993]. Since invertebrate mapping schemes typically have only a limited number of collaborating volunteers, a larger part of the field work would have to be done by [expensive] professional workers.
Furthermore, the more fluctuating numbers in invertebrate populations ren-

ders interpretation of differences in abundance more difficult than in long-living vertebrates like birds [Thomas 1994]. Since only few mapping schemes can rely on as large a numbers of volunteers as for birds or vascular plants, other [especially invertebrate] mapping schemes should consider the desired information on the one hand and the possible application of the information on the other. This could result in a joint venture of different actors [e.g., conservation practitioners, research institutions, volunteer working groups] and the mapping of a limited number of species and/or sites after which modelling can be applied to extrapolate the results to the whole focal region [cf. **Chapters 4-5**].

Red Lists

Red Lists in Flanders, the Netherlands and Germany are compiled using two criteria: a rarity criterion indicating the actual geographical extent of the species and a trend criterion indicating the change in distribution area between two periods [**Chapter 2**]. But, calculating changes in distribution area using coarse grain distribution data [5x5 km squares] can strongly underestimate the decline of species [León-Cortés *et al.* 1999; Thomas & Abery 1995]. This is not necessarily the case for very rare species, but is particularly true for intermediately rare or even common species [Thomas & Kunin 1999; Van Dyck 2000]. Cowley *et al.* [1999] have shown, by comparing the extent of historical and actual habitat patch sizes, that intermediately rare butterflies have declined at the same rate as many of the threatened species. Such declines remain unnoticed in the present Red Lists. Therefore, we need data on species, or at least on a selection of them such as species from the categories ‘Critically endangered’ and ‘Endangered’ and ‘Vulnerable’, completed with a number of intermediately rare and/or fairly sedentary species [Thomas 2000], that can keep a finger on the pulse on a year-to-year basis. Ideally, a system in which a subset of populations of all Red List species, plus a selection of intermediately rare species is monitored yearly, would make the conservation of [threatened] species much more efficient and pro-active than is currently the case. The minimal number of counting points for monitoring can be determined on the basis of model simulations with the extensive Dutch butterfly monitoring scheme [van Swaay *et al.* 2002]. A minimal monitoring network for butterflies, for example, could consist of about 50 transects that are equally spread over the different ecological regions and over the different biotope types [van Strien *et al.* 1997]. Examples of such monitoring schemes can be found for butterflies in Great Britain [Pollard & Yates 1993] and in the Netherlands [van Swaay *et al.* 2002; van Swaay & Plate 2002] and for birds in Flanders [Anselin *et al.* 2003] and Great Britain [Common Bird Census of the British Trust for Ornithology, http://www.bto.org/survey/cbc.htm]. Monitoring schemes make it possible to calculate long and short term trends of all species, including Red List species [Balmford *et al.* 2003]. Such 'long'-term time series are already available in The Netherlands for a variety of taxa [birds, dragonflies, reptiles, amphibians, fish, etc.] and regular reports clearly indicate trends in the distribution or the numbers of 'indicator' species [van Duuren *et al.* 2003]. In Flanders, comparable long-term time series are only available for waterbirds [Devos *et al*. 1997, 1998, 2001] and the development of similar time series for other taxa would be a valuable tool for the two-yearly Nature Reports in Flanders [Dumortier *et al.* 2003; Kuijken 1999; Kuijken *et al.* 2001].

A preliminary Red List for ecotopes in Flanders has been compiled recently [Van Landuyt 2002] and again demonstrates that the threat status of the biotope type is not necessarily a good surrogate for the degree of threat of the associated species. This, once again, stresses the importance of incorporating species into site-based conservancy policies. Although for some

biotope types, the degree of concordance in decline between the biotope type and the typical species is high [e.g., wet and dry heathland], other biotope types do show fairly large discrepancies [e.g., the biotope type 'deciduous woodland' is not threatened in Flanders, but many of the typical woodland butterfly species are extinct or threatened in Flanders – Gorissen *et al.* in press]. For practical species conservation, an important distinction has to be made between the biotope and the habitat of a species [Dennis *et al.* 2003]. Within a biotope [e.g., dry heathland] particular habitats [e.g., early succession dry heathland with open sandy patches and lichens] probably declined more strongly than the biotope as a whole [Vanreusel *et al.* 2002].
In Flanders, Red Lists are available for a fairly large numbers of taxonomic groups compared to other countries or regions [Table 1.1]. Uniformity in the methodology to compile Red Lists facilitates the comparison of threat statuses among different taxonomic groups [**Chapter 2**]. However, due to temporal and spatial differences [**Chapter 5**], the comparison of threat statuses among species from different taxonomic groups remains difficult.
Furthermore, the frequency with which Red Lists are published differs among taxonomic groups. In Flanders, it has been proposed to review Red Lists every 10 years [Maes *et al.* 1995]. Such relatively long time periods are often necessary because actual Red Lists are based on mapping schemes for the whole Flemish territory. The disadvantage of this approach, however, is that it is only capable of assessing the extinction of species in a next Red List. A pro-active nature conservation policy should be able to take measures before species go extinct and monitoring of populations of threatened species therefore seems the most appropriate technique.
In the Netherlands, Red Lists are officialized lists of species for which the biotope needs to be protected [Stroo 2003]. Such an approach is far more pro-active than the protection of the species themselves, i.e., a ban on capturing, selling, collecting a [threatened] species. Species protection arose

mainly because collecting was seen as a major cause of threat to rare species. Although this can be true in very small populations, it has become clear that collecting is not the major cause of recent species declines [Thomas 1983; van Swaay & Warren 1999]. Species protection *sensu stricto* should, therefore, always be an integrated part of a much broader array of protection measures [e.g., adequate site selection and nature management for protected areas, improving environmental quality ...].

The IUCN has recently proposed the use of uniform criteria for compiling Red Lists both on a global and on a local level [IUCN 2001]. These criteria have their origin in the compilation of global Red Lists and the IUCN now recommends their use on the local level as well. Five major criteria are applied in the IUCN criteria [between brackets the criteria for classification as 'Critically endangered' according to the IUCN]: A] population size reduction [>80% in the last 10 years], B] small distribution area [<100 km^2], C] small population size [<250 mature individuals] and decline in distribution extent [>25%], D] extremely small population [<50 mature individuals] and E] probability of extinction [population viability analysis shows that the probability is >50% in the next 10 years] [Gärdenfors *et al.* 2001]. However, these new regional IUCN criteria are difficult to apply for the compilation of Red Lists for invertebrates and other, more inconspicuous taxonomic groups [Hallingbäck *et al.* 1995]. Population sizes [the total number of individuals, criteria A, C, D and E] are almost impossible to obtain for invertebrates and can fluctuate strongly among years. The only applicable criterion for invertebrates is the distribution area criterion B, but classifying species on the single criterion of rarity does not allow for an appropriate prioritisation. By using changes in distribution area instead of changes in population size [both are strongly correlated; Warren *et al.* 2001], it is possible to interpret criterion A of the IUCN guidelines as a measure of decline. For small regions such as Flanders, however, the figures to determine rarity [expressed as the distribution area] used in criterion B of the IUCN guidelines are not appropriate and will have to be related to the size of the focal region [Hartley & Kunin 2003; **Chapter 2**]. Additionally, Hartley & Kunin [2003] propose the use of a multiscale measure of rarity [100 km^2, 10 km^2, 1 km^2, ha] to extrapolate fine-scale measurements of area of occupancy. This promising method needs to be tested on those Flemish databases that have a sufficient number of fine-scale distribution data [minimum 1 km^2]. For some species for which sufficient data on population sizes are available [e.g., rare breeding birds, *Maculinea alcon* ...], the new IUCN criteria could be applied to assess their Red List status. Ideally, such population size information should be gathered for more widely spread, and even for common species as well in order to estimate to what extent different rarity estimates influence the classification of species with the IUCN criteria. The status obtained this way can then be compared with that obtained by the presently applied method in Flanders. This would allow to test whether the more easily applicable actual Red List classification method is sufficiently robust to be used for nature conservation purposes.

Species action plans

Species action plans are scientifically underpinned plans with specific policy or management measures for a threatened species. They are useful tools to adapt management measures and/or management regimes inside, but also outside nature reserves in order to restore specific components of biodiversity [Van Dyck *et al.* 2004]. Ideally, the compilation of species action plans would be easier, quicker and cheaper when species are not yet at the brink of extinction [Thomas 1991], but both financially and practically, this is, at present, not a realistic approach.
Until now, in Flanders, no criteria exist for prioritising the compilation of

species action plans. This resulted in rather arbitrary choices of species for which such plans have been compiled [Table 1.2]. Red Lists could be used as a basis for the compilation of species action plans, but other approaches are possible: combining European nature conservancy Directives [Bird Directive annex I or Habitat Directive annexes II or IV, Bern Convention annexes I or II] with Flemish Red Lists ['Critically endangered' or 'Endangered' species] can be a good starting point to rank species that are in need of a species-specific conservation approach. Applying such combined criteria for species in Flanders for which the Red List categories have been assessed, results in Table 9.1.

Due to the strong taxonomic bias towards vertebrates and vascular plants in the European conservancy directives, only three invertebrates appear in Table 9.1. This bias is due to the available information at the time of compilation of the different annexes of the Habitat Directive and the Bern convention. But, both for vertebrates and invertebrates the incorporation of species in the different annexes was based on 'best professional judgements' and not on numerical criteria [cf. IUCN 2001; Stroo 2003]. The recently published Red List of European butterflies [van Swaay & Warren 1999], for example, has shown that some of the species on the annexes are not threatened and that, on the contrary, some of the threatened species are not listed in the annexes of European directives. In order to compensate for the bias towards vertebrates and incorporate a greater variety of [especially invertebrate] species in species action planning, other approaches can be applied as well. In **Chapter 3** we propose a method that calculates changes in distribution area using coarse grain atlas data. This technique can be applied to all existing distribution data bases to detect both the taxonomic group that declined most strongly, but also to compare declines of individual species across taxonomic groups. Applying this method, for example, to a selection of three invertebrate groups [dragonflies, butterflies and grasshoppers] results in a ranking of the most threatened species [Red List categories 'Critically endangered' and 'Endangered'] independent of its taxonomic affiliations [Table 9.2]. This ranking confirms that butterflies are the most threatened of these three groups in Flanders [Maes & Van Dyck 2001]. Such rankings [applied to a larger number of taxonomic groups] could be a useful tool to determine priorities in the compilation of species action plans. Other authors [e.g., Telfer *et al*. 2002] have proposed different techniques to correct for mapping intensity when calculating trends using distribution data but usually only allow to compare trends of the taxonomic group as a whole or of species within a particular taxonomic group. Furthermore, threat status should not be the single criterion in the prioritisation of species action plans. Two additional criteria are equally important: priority should be given to species that do not benefit sufficiently from regular management measures of their biotopes within nature reserves on the one hand and to species that are mainly distributed outside nature reserves on the other [Van Dyck *et al*. 2004].

Table 9.1. Priority list for species action plans in Flanders based on European and regional threat and conservation status. For species marked with * a species action plan has already been compiled. BC = Bern Convention annex; BD = Bird Directive, HD = Habitat Directive annex, RLF = Red List Category Flanders [CR = Critically endangered, EN = Endangered].

Taxonomic group	Species	Dutch Name	BC	BD	HD	RLF
Plants						
Vascular plants	*Apium repens*	Kruipend moerasscherm	I	-	II	CR
Vascular plants	*Liparis loeselii*	Groenknolorchis	I	-	II	CR
Vertebrates						
Herpetofauna	*Alytes obstetricans**	Vroedmeesterpad	II	-	IV	EN
Herpetofauna	*Hyla arborea**	Boomkikker	II	-	IV	CR
Herpetofauna	*Pelobates fuscus*	Knoflookpad	II	-	IV	EN
Birds	*Botaurus stellaris*	Roerdomp	II	I	-	CR
Birds	*Caprimulgus europaeus*	Nachtzwaluw	II	I	-	EN
Birds	*Charadrius alexandrinus*	Strandplevier	II	-	-	CR
Birds	*Circus pygargus*	Grauwe kiekendief	III	I	-	CR
Birds	*Crex crex*	Kwartelkoning	II	I	-	CR
Birds	*Emberiza citrinella*	Geelgors	II	-	-	EN
Birds	*Emberiza hortulana*	Ortolaan	III	I	-	CR
Birds	*Ixobrychus minutus*	Woudaapje	II	I	-	CR
Birds	*Lanius collurio*	Grauwe klauwier	II	I	-	CR
Birds	*Lanius excubitor*	Klapekster	II	-	-	CR
Birds	*Oenanthe oenanthe*	Tapuit	II	-	-	CR
Birds	*Porzana porzana*	Porseleinhoen	II	I	-	EN
Birds	*Riparia riparia*	Oeverzwaluw	II	-	-	EN
Birds	*Saxicola rubetra*	Paapje	II	-	-	CR
Birds	*Saxicola torquata*	Roodborsttapuit	II	-	-	EN
Birds	*Sterna albifrons*	Dwergstern	II	I	-	CR
Birds	*Sterna hirundo*	Visdief	II	I	-	EN
Birds	*Sterna sandvicensis*	Grote stern	II	I	-	CR
Birds	*Tetrao tetrix*	Korhoen	III	I	-	CR
Mammals	*Cricetus cricetus**	Hamster	II	-	IV	CR
Mammals	*Muscardinus avellanarius*	Hazelmuis	III	-	IV	EN
Mammals	*Myotis bechsteinii**	Bechsteins vleermuis	II	-	II+IV	CR
Mammals	*Myotis brandtii**	Brandt's vleermuis	II	-	IV	EN
Mammals	*Myotis dasycneme**	Meervleermuis	II	-	II+IV	EN

Taxonomic group	Species	Dutch Name	BC	BD	HD	RLF
Mammals	*Myotis emarginatus**	Ingekorven vleermuis	II	-	II+IV	CR
Mammals	*Myotis myotis**	Vale vleermuis	II	-	II+IV	CR
Mammals	*Nyctalus leisleri**	Bosvleermuis	II	-	IV	CR
Mammals	*Phoca vitulina*	Gewone zeehond	III	-	II	CR
Mammals	*Phocoena phocoena*	Bruinvis	II	-	II+IV	CR
Mammals	*Plecotus austriacus**	Grijze grootoorvleermuis	II	-	IV	EN
Invertebrates						
Land snails	*Vertigo angustior*	Nauwe korfslak	II	-	II	CR
Land snails	*Vertigo moulinsiana*	Zeggekorfslak	II	-	II	CR
Dragonflies	*Leucorrhinia pectoralis*	Gevlekte witsnuitlibel	II	-	II+IV	RE

Table 9.2. Ranking of species [in decreasing order of decline] from three invertebrate taxonomic groups [dragonflies, grasshoppers and butterflies] according to the method proposed in Chapter 3. * = species action plans already compiled [Vanreusel *et al.* 2000]. RLF = Red List Category Flanders [CR = Critically endangered, EN = Endagered, EX = Extinct].

Species	Dutch name	Taxonomic group	RLF
Polyommatus semiargus	Klaverblauwtje	Butterflies	CR
Melitaea cinxia	Veldparelmoervlinder	Butterflies	CR
Pyrgus malvae	Aardbeivlinder	Butterflies	EN
Issoria lathonia	Kleine parelmoervlinder	Butterflies	CR
Leptidea sinapis	Boswitje	Butterflies	CR
Tetrix tenuicornis	Kalkdoorntje	Grasshoppers	EN
Hesperia comma	Kommavlinder	Butterflies	EN
*Maculinea alcon**	Gentiaanblauwtje	Butterflies	EN
Coenagrion hastulatum	Speerwaterjuffer	Dragonflies	EN
Leucorrhina pectoralis	Gevlekte witsnuitlibel	Dragonflies	EX
Apatura iris	Grote weerschijnvlinder	Butterflies	EN
Aeshna isosceles	Vroege glazenmaker	Dragonflies	CR
Libellula fulva	Bruine korenbout	Dragonflies	CR
Platycleis albopunctata	Duinsabelsprinkhaan	Grasshoppers	EN
Gryllotalpa gryllotalpa	Veenmol	Grasshoppers	EN

Not only is the choice of species very arbitrary, the content of the different existing species action plans is also very heterogeneous: some are based on original ecological research and give a detailed overview of actions and further research to be undertaken [Vanreusel *et al.* 2000]; others compile existing ecological information and are vague in their recommendations. Future species action plans should therefore have a more uniform approach and proposed actions should be underpinned with the best available knowledge on the species [Pullin 2002a; Pullin & Knight 2003]. Furthermore, clear objectives, time schedules and priorities should be set in order to evaluate the proposed measures quantitatively and, if necessary, adjust them accordingly [Ruckelshaus *et al.* 2003]. Species action plans should consist of two major parts: a general part compiling existing, but also missing, information on distribution and autecology and a more specific part where detailed management and policy measures are listed per site [both for actual, historical and potential reintroduction sites; cf. **Chapter 7**; Vanreusel *et al.* 2000]. A clear communication plan towards both managers and policy makers, should be an explicit part of every future species action plan [Cort 1996; Foin *et al.* 1998; **Chapter 7**]: local nature managers will primarily need fine-scale maps with detailed descriptions of the necessary management measures, while policy makers will have to be informed about necessary funding for site acquisition and/or monitoring, protecting the focal species legally, preparing a legal frame for a reintroduction scheme, inform others on the necessity and consequences of the species action plan, etc. [Pullin 2002a]. A further necessity in future species action plans is the specification of operational goals [the desired number of populations/individuals or sites that should be obtained within a defined time period; Ruckelshaus *et al.* 2003]. Without such operational targets, the effectiveness of species action plans cannot be estimated. At present, most compiled species action plans in Flanders remain academic exercises because the, albeit relevant, information is not presented in a workable format on the one hand [e.g., practical field guidelines for managers] or because the obtained and/or compiled information is irrelevant for the conservation of the species on the other [e.g., distribution data are gathered on the wrong scale]. This can result in an inadequate implementation of proposed conservation measures. Two major causes can be indicated for the gap between action plans and their implementation in the field in Flanders. A first drawback is the lack of a follow-up commission, that can verify if suggested measures are effectively executed or that can support local authorities or conservation practitioners to do so. Such follow-up commissions could be installed for every species action plan, but a more integrated and more permanent species action plan commission seems more appropriate. A follow-up commission should be composed of the different actors involved [scientists, practitioners and policy makers] and should meet at yearly intervals to evaluate progress in the implementation of the different species action plans. The scientific units of the Flemish government [the Institute of Nature Conservation and/or the Institute for Forestry and Game Management] are the most appropriate administrative coordinators of such follow-up commissions and should, at the same time, provide information on priorities in compiling species action plans. A second problem is the absence of a standardised monitoring scheme to estimate the consequences of the action plan for the actual conservation of the species in question. Some plans, however, do get implemented because of local initiatives [volunteers, wardens] but the authorities should invest more in the implementation of species action plans if they are to be effective [both cost-effective and effective in conserving the species]. Proposed measures in species action plans are often too vague to be implemented and a more precise terminology and detailed maps with management instruction should be incorporated to facilitate the communication with and practical relevance for the end users [Foin *et al.* 1998;

Vanreusel *et al.* 2000]. Many of the existing species action plans in Flanders are compiled by experts by order of the nature conservancy authorities. By specifying clearly what a species action plan should contain and what information is absolutely necessary for the conservation of the species in question [e.g., distribution, ecology, mobility, behaviour], the authorities themselves could also contribute to a greater uniformity in and a wider applicability of future species action plans. The species action plan for the Alcon Blue butterfly *Maculinea alcon* was compiled in close collaboration with several professional and volunteer conservation practitioners [Vanreusel *et al.* 2000]. In this plan, detailed maps indicated where to implement what kind of management measures; furthermore, the extent of these measures and the time period in which they had to be executed were described in detail. The implementation of such detailed management proposals and a clear communication was very successful in two of the larger populations of the Alcon Blue butterfly *Maculinea alcon* [Zwarte Beek, Koersel-Beringen and Hageven, Neerpelt] where the species increased both in extent [due to the enlargement of the habitat patches] and in numbers [due to the exclusion of cattle in the egg-laying period of the butterfly].
Research on threatened [invertebrate] species, used to compile local or national species action plans, is often done in two of our neighbouring countries, i.e., Great Britain [see lists of species action plans for butterflies on http://www.butterfly-conservation.org] and The Netherlands [Ministerie van Landbouw Natuurbeheer en Visserij 1990; the Glanville Fritillary *Melitaea cinxia*: Wallis de Vries 2001a; the Alcon Blue butterfly *Maculinea alcon*: Wallis de Vries 2003]. Flanders should make use of recommendations made there. But, the information and suggestions made for species in other countries are not necessarily applicable here and some of the published information can even appear to be wrong, leading to inappropriate measures in Flanders [Pullin & Knight 2003]. Examples of such misleading

or ‘false’ information in specialist literature were encountered when compiling the species action plan for the Alcon Blue butterfly *Maculinea alcon* [Vanreusel *et al.* 2000; **Chapter 7**]. A first example regards the statement that the Alcon Blue butterfly *Maculinea alcon* can survive in small habitat patches as long as the perennial host plant [Marsh Gentian *Gentiana pneumonanthe*] remains present [Bink 1992]; clearly wrong advice was found about the fact that grazers [cattle, horses or sheep] do not eat the Marsh Gentian *Gentiana pneumonanthe* [the butterfly’s host plant] because of the bitter taste. **Chapter 7** clearly indicates that small populations have a greater extinction risk than larger ones and our own observations showed that cattle used in grazing management eat a considerable number of Marsh Gentian *Gentiana pneumonanthe* plants [on which eggs were present]. Uncritically applying such information can cause the extinction of the species in small local populations, because reserve managers solely rely on the presence of the conspicuous host plants; furthermore, grazing is one of the best management measures for the conservation of wet heathland [the biotope of the Alcon Blue butterfly *Maculinea alcon*] on the long-term, but the incorporation of some short comments on the possible impact of grazers on the number of eggs is highly recommended and can increase the survival potentials of local populations of the Alcon Blue butterfly *Maculinea alcon* considerably. Including behavioural aspects of the Alcon Blue butterfly *Maculinea alcon* into the research for the species action plan further revealed that females of the Alcon Blue butterfly *Maculinea alcon* preferred to oviposit on host plants that are in the immediate vicinity of *Myrmica* ant nests [Van Dyck *et al.* 2000]. This was also contrasting with existing literature that reported that oviposition was random in all *Maculinea* species [Fiedler 1991] although further research is needed to detect whether this preference is causal or correlative. Therefore, specialist literature always needs to be verified relative to local conditions. One of the best known examples of how detailed autecological research resulted in the successful conservation of an invertebrate species is the one on the Large Blue butterfly *Maculinea arion* in England [Thomas 1991]. *Maculinea arion* is a myrmecophilous species that uses *Thymus* spp. as host plants and *Myrmica sabuleti* as host ant. The key-factor for the successful reintroduction was microclimate [related to vegetation height]: in populations in southern France, the Large Blue butterfly *Maculinea arion* [and especially the host ant *Myrmica sabuleti*] lives in vegetations of 15-50 cm; a similar microclimate in Britain was only obtained on southerly slopes with a vegetation height of <3 cm [Thomas 1993]. Former sites of the Large Blue butterfly *Maculinea arion* were managed according to this new information and the subsequent reintroduction was very successful [Thomas 1995].

Some of the species action plans suggest the reintroduction of threatened species as an ultimate tool for their conservation. But, an important shortcoming in the use of such 'intensive care' management as part of a species action plan is the lack of a clear reintroduction policy in Flanders [both by the local authorities and by the NGO's] on the one hand, and of a legal frame on the other [Ulenaers 1995; Van Den Berge *et al.* 1995]. Most reintroductions are now illegally done by well-intentioned volunteers without any preliminary scientific research and often without subsequent monitoring of the reintroduced populations [e.g., beavers in the valley of the Dyle]. Several illegal reintroductions fail to conserve species in the long-term because of an insufficient number of reintroduced individuals, an insufficient knowledge of the specific habitat requirements, inappropriate management schemes on the site of reintroduction, etc. [Dempster & Hall 1980; Kuussaari *et al.* 1996; Oates & Warren 1990]. Among others, Munguira & Martín [1999], Oates & Warren [1990], Thomas [1995] and Wynhoff [2001] have stressed the importance of using sound scientific data and insights for species reintroductions. Furthermore, [experimental] reintroductions can help to identify underlying reasons for the decline and subsequent extinction of species and can prove invaluable as natural experiments on population genetics, genetic drift, founder effects and the effects of isolation [Oates & Warren 1990]. A clear and open-minded policy on reintroductions as a conservation tool is therefore urgently needed. A first step is to clarify definitions on [re]introductions and translocations: a *reintroduction* is defined as the release of species in a site where it went extinct in the past after the cessation of the causes that led to the extinction; a *translocation* is defined as the transfer of species to other parts of the same area as a risk-spreading strategy or because of limiting factors preventing a spontaneous colonisation within the area [IUCN 1987]. A first instigation of a critical but scientifically underpinned reintroduction proposal in Flanders is done for the Alcon Blue butterfly *Maculinea alcon* [Vanreusel *et al.* 2000; Vanreusel *et al.* 2002]. Combining propositions of these authors together with the ones of other experts [e.g., Kuijken & De Blust 2002; New 1995b; Thomas 1995; Wynhoff 2001], the following protocol for reintroductions in Flanders is suggested:

1 Historic-ecological background research, during which the former distribution of the species should be investigated together with the reasons for its extinction;

2 The implementation of particular management measures in order to increase the extent and the quality of suitable habitat patches;

3 In-depth research on possible source populations;

4 Compilation of a detailed plan of execution [methods, numbers and life stage to be reintroduced, monitoring plan, etc.];

5 The actual reintroduction;

6 Follow-up of reintroduction, monitoring and possible adjustments to management measures or local site acquisition.

Ideally, species should be reintroduced in sites where they were historically present [Oates & Warren 1990]. But, knowledge on the historical distribution of an endangered species is often only available on a coarse scale [e.g., mapping grid cells of 5 x 5 km or municipalities] and very rarely on a detailed site-level. It is, therefore, appropriate to adopt the definition of the IUCN [1987] that states that a reintroduction should not necessarily be into a historically occupied site, but into "... *a part of its native range* ...". This increases the number of possible reintroduction sites considerably and permits reintroductions into suitable sites where the species was present historically, but remained undocumented [i.e., no historical data are present in the mapping scheme data base].

Evaluating and/or planning conservation measures using species information

In Flanders [but also elsewhere], the implementation of evidence-based conservation biology on conservation practice is still limited compared to the more widespread use of experience- and tradition-based management [cf. Pullin & Knight 2003; Salafsky *et al.* 2002]. This leads to the application of many management measures solely on cultural or historical grounds [e.g., 'biodiversity was high when in the beginning of the 20th century, grasslands were only mown once']. There is a considerable risk that conservation practitioners will consider management techniques as a target on their own and not as tools to improve or maintain conditions for local biodiversity under current landscape and environmental conditions.
Particularly for the management of semi-natural, traditionally managed biotopes [like heathlands or nutrient poor grasslands in Europe], there is a growing awakening of the need to take into account species-specific requirements [that are not necessarily guaranteed by restoring traditional management techniques; Pullin & Knight 2001, 2003; **Chapters 6-8**]. In fact, traditional agricultural techniques were never intended or developed to con-

serve biodiversity. Moreover, their present-day impact most probably differs from that in the past because the environmental quality [nitrogen deposition] and the landscape context [fragmentation] have changed considerably [Thomas *et al.* 1998a]. Therefore, it is highly relevant to draw species information with respect to habitat quality, quantity and geometry into the focus of management. Hence, effects of management should be evaluated by using existing or by gathering appropriate species information such as population densities [what are normal densities in unmanaged situations?], habitat specificity [is the species confined to specific habitats in the managed biotope?], sensitivity to environmental stressors [are other factors than management more important for the species survival?], etc. [Niemi *et al.* 1997]. For example, grazing is a commonly applied management measure in many nature reserves in Flanders and is usually promoted on historical and/or practical grounds and not necessarily on evidence-based knowledge. Several scientific studies suggest that grazing does not necessarily result in a higher species diversity or in higher abundances of the target species [e.g., spiders - Bell *et al.* 2001; Zulka *et al.* 1997; butterflies and Hymenoptera - Kruess & Tscharntke 2002; bumblebees - Carvell 2002]. They do agree, however, that low intensity grazing is preferable to intensive grazing. Nature management plans in Flanders should make more use of experiment-based research by using species in evaluating and/or monitoring the results of different management measures in the same biotope type. Most of the present-day management plans do not incorporate guidelines based on experimental evidence [Pullin & Knight 2003]. In Flanders however, the limited area of nature reserves and hence of particular biotopes and spatial management units, may hold the risk of pseudo-replicates for such experimental set-ups as different plots of the same current management regime within and among reserves may vary considerably in management history which confounds comparisons. However, such experimental approaches applied in a selection of nature reserves, permit to evaluate whether certain measures are more effective compared to others and allow for a more objective comparison with preset goals. Results obtained through such experimental management approaches should consequently be appropriately communicated to managers in other reserves; they can then decide whether local management should be adapted accordingly. Although taxonomically biased towards the well-known groups such as vascular plants, birds and mammals, species lists exist for many nature reserves or other conservation areas and are often incorporated in annual reports. The utility of such species lists, however, could be greatly improved by adding ecologically significant species-specific information [e.g., on habitat use, spatial scale of mobility, rarity, etc.; cf. the Conservation Management System in Great Britain, http://www.cmsp.co.uk/]. Futhermore, such species lists are useful to detect possible 'missing' species compared to the expected species pool based on available biotope types and regional species lists [Speight & Castella 2001]. The incorporation of annotated lists of characteristic species for the different biotope types [so called 'Natuurtypen', e.g., Vandenbussche *et al.* 2002] makes such analyses possible taking differences in ecological regions in Flanders into account. The information content [trophic level, scale dependency, etc.] of 'missing' species can be applied to take specific actions concerning habitat quality [through management measures] and habitat quantity and/or configuration [through the aquisition of surrounding suitable biotopes]. An integrated logbook for nature reserves [preferably in a geographic information system] with applied management measures, species lists and autecological information could facilitate a scientific review and evaluation of management measures considerably [Clark *et al.* 2002]. Such integrated logbooks can contribute to the development of so called Decision Support Systems [Garcia & Armbruster 1997] that are presently being developed for some of the larger nature reserves in Flanders [Research Institute voor KennisSystemen 2002].

The Nature Decree [21 October 1997] obliges private nature organisations to monitor a list of focal species. This list, however, was not compiled using scientific criteria [Van Dyck *et al.* 1999] resulting in a list of species of which some have never been indigenous in Flanders, while others are already extinct or have become extremely rare [e.g., none of the listed dragonfly species is actually present in Flanders; Van Olmen *et al.* 2000]. The list of focal species is an example of the hitherto poorly thought-out use of species in nature conservation in Flanders. Criteria for the selection of focal species depend on what they should indicate [Caro & O'Doherty 1999; Hilty & Merenlender 2000; Lambeck 1997; Mc Geoch 1998; Noss 1990]. If a list of species is needed to evaluate nature conservancy policy in Flanders, the number of Red List species, for example, would be a possible 'indicator' [Noss 1990]. But, if practitioners want to evaluate the suitability of an implemented management measure, the use of other criteria for species selection may result in a different suite of species. However, indicator species [or other surrogate measures such as diversity, i.e. the total species richness] should always meet minimum requirements on knowledge about taxonomy, ecology and distribution [Hilty & Merenlender 2000; Noss 1990]. The data on focal species in nature reserves have to be centralised by the Institute of Nature Conservation, but a proper protocol or format to deliver them is lacking. This makes sound analyses difficult, if not impossible [De Bruyn 2003]. Uniformity in collecting techniques, formats and computerisation of these data [for example, how should the information be collected and what kind of information is needed exactly] would greatly facilitate the treatment of the data. These analyses could subsequently indicate the most urgent conservation [e.g., species actions plan] or management actions for threatened species within nature reserves. Additionally, adding [semi-]quantitative data

on population size [order of magnitude estimated in categories] and supplementary data on management measures would greatly increase the value of this monitoring obligation. Finally, the obligation to monitor focal species should not only apply to private nature organisations but should be extended to public nature reserves as well [Demeulenaere *et al.* 2002].
A seldomly used information source in nature conservation in Flanders, but also elsewhere, is behavioural research [Caro 1998], especially on invertebrates. However, the results of such research can be applied in various nature conservation applications such as the optimal design of corridors [Haddad 1999; Haddad & Baum 1999; Haddad *et al.* 2003; Simberloff *et al.* 1992; release experiments in **Chapter 7**], the impact of biotope boundaries on emigration [Kuussaari *et al.* 1996; Schultz 1998; Schultz & Crone 2001; Crone & Schultz 2003] or the effect of the intermediate landscape on dispersal [Adriaensen *et al.* 2003; Chardon *et al.* 2003; Merckx *et al.* 2003; Ricketts 2001]. It was assumed that the restoration or maintenance of corridors could counteract the increasing extinction rates of relatively sedentary species in fragmented landscapes [Wilson & Willis 1975]. Despite very little empirical evidence for this assumption, the concept of corridors for wildlife is now widely used in many land development and nature restoration projects [Simberloff & Cox 1987; Simberloff *et al.* 1992; Sutcliffe *et al.* 2003].
The term 'corridor' can be interpreted in two different ways: 1] specific connections between two sites [i.e., biotope types] or 2] spatial areas with a large number of connecting elements [e.g., hedgerows]. In Flanders, the provincial authorities have to specify so called corridor zones for the Flemish Ecological Network, but no criteria [or species] or any other protocol are given as guidelines for their designation. Since no such thing as a 'universal corridor' exists, corridor design [length, width, etc.] will have to be based on a selection of model organisms [Chardon *et al.* 2003; Verbeylen *et al.* 2003]. Some species indeed make use of linear elements to move

through the landscape [e.g., *Metrioptera roeseli*; Berggren *et al.* 2002], while others do not always use the landscape as expected: several authors [together with the release experiments with the Alcon Blue butterfly *Maculinea alcon*; **Chapter 7**] have shown that species do not necessarily use linear landscape elements [woodland rides, hedgerows, road verges, etc.] for movements between two similar biotopes. Behavioural observations on the Fender's Blue butterfly *Icaricia icarioides fenderi* within and outside suitable habitat revealed that a network of stepping stones between existing patches would be more effective for the colonization of new patches than corridors [Schultz 1998]. In this respect, it has also been hypothesized that a landscape with a network of hedgerows may encourage woodland butterflies [e.g., the Speckled Wood *Pararge aegeria*] to cross a woodland boundary and that a network of hedgerows would rather function according to a stepping stone principle than to corridors *sensu stricto* [Merckx *et al.* 2003]. Additionally, behavioural observations on the Glanville fritillary *Melitaea cinxia* [Kuussaari *et al.* 1996] and on the Fender's Blue butterfly *Icaricia icarioides fenderi* [Schultz & Crone 2001] have shown that emigration is much lower from patches surrounded by a distinct physical barrier [forest, dense tree row, etc.]. Changing the design of patch boundaries of isolated populations [e.g., by planting tree rows] could prevent individuals from emigrating into an unsuitable landscape matrix [as suggested for the isolated population of the Alcon Blue butterfly *Maculinea alcon* in Houthalen-Helchteren; Vanreusel *et al.* 2000].

Parallel to a centralised data base with species distribution records, there is a similar need for a data base that indicates existing [and missing] ecological information of species in Flanders. However, a rigorous screening on the scientific correctness of the stored information or the applicability of foreign information in Flanders is essential and is a task for statutory bodies such as the Institute of Nature Conservation or the Institute for Forestry and Game Management [cf. Meffe *et al.* 1998; Pullin & Knight 2003]. This data base should contain information on detailed habitat use [preferably resource-based - Dennis *et al.* 2003; Speight & Castella 2001], mobility, sensitivity for environmental stressors, trophic level, host plant, relations with other species, behaviour, etc. Centralisation of both the distribution and the ecological data base would greatly facilitate analyses like the ones performed in **Chapters 3-5**. Making this data base accessible [via an interactive website] to possible end users such as wardens and policy makers, would certainly increase the day-to-day use of species information in nature conservation on condition that the information provided is in an applicable format. Since such detailed information is particularly scarce for invertebrates, the gathering of autecological data for the invertebrate taxonomic groups that are already being mapped in Flanders [e.g., ants, butterflies, dragonflies, grasshoppers, spiders, carabid beetles, ladybirds and hoverflies] should therefore be structurally supported by the regional authorities. The same holds true for lower plants such as fungi, bryophytes and lichens.

Applying multi-species approaches in Flanders

The large number of [threatened] species does not allow to gather and use information of all species simultaneously. Therefore, nature conservation often applies the short-cut concept of indicator species or groups [Landres *et al.* 1988]. Since more information is available for vertebrate species and for vascular plants, these taxonomic groups usually serve as guidelines whenever species are used for planning and/or evaluating management measures or site selection [Cabeza & Moilanen 2001; Simberloff 1998]. Recently, several authors have shown that the use of one taxonomic group or a single species does not necessarily result in the conservation of other species or taxonomic groups as well [Landres *et al.* 1988; Prendergast *et al.*

1993a; Simberloff 1998; van Jaarsveld *et al.* 1998]. Different authors therefore suggest the use of so called multi-species approaches where a carefully selected group of taxonomically and ecologically different species provides complementary information on spatial, quantitative and qualitative aspects of conservation areas, management measures or nature conservancy policies [Coppolillo *et al.* 2004; Fleishman *et al.* 2000; Jeanneret *et al.* 2003; Kotze & Samways 1999; Root *et al.* 2003; Vanderklift *et al.* 1998; Van Dyck *et al.* 1999]. But, in order to make them applicable in nature conservation and usable by conservation practitioners, multi-species approaches should meet a number of minimum criteria [Deem *et al.* 2001; Pullin & Knight 2001; Robertson & Hull 2001]: the species of the multi-species group should, preferably, be easily recognizable by non-experts and should, at the same time, provide information on other threatened or typical species and on habitat quality. The information content of multi-species groups can be explicitly used in the evaluation or the set up of conservation actions [Mc Geoch 1998]. Multi-species approaches have the clear benefit of forcing conservationists 'to cross' taxonomic boundaries and hence to explicitly take into account different requirements and different scales that are relevant for different components of biodiversity. The use of multi-species approaches represents a continuous process rather than a one-off operation [Fleishman *et al.* 2001b] and stresses the necessity of gathering information on a structured and on a long-term basis [Mc Geoch 1998].
However, the use of a group of taxonomically and ecologically different species for the planning or evaluation of conservation measures always remains a simplification following from inevitable pragmatism for conservation practice [Jeanneret *et al.* 2003].
Multi-species approaches can be used in several conservation applications such as the description of biotope types, the evaluation of habitat quality, assessing the impact of nature management, nature restoration or land

development projects, the selection of sites for the Flemish Ecological Network, etc. In **Chapter 8**, it has been shown that both a threatened species [in our case the Alcon Blue butterfly *Maculinea alcon*] and a multi-species group have capacities as indicators for the characteristic species diversity of wet heathland, a threatened biotope type in Europe. In the case of wet heathland, only the multi-species group of nine 'indicator' species appeared to be a good indicator of habitat heterogeneity, whereas the Alcon Blue butterfly *Maculinea alcon* alone failed to do so. Signals can indeed be picked up through the use of a multi-species umbrella, but further research usually remains necessary to determine whether species' reactions are causally linked to environmental changes or not. An advantage of the multi-species approach proposed for wet heathlands in Flanders, however, is that non-experts can evaluate habitat heterogeneity using the multi-species umbrella in a fairly large region and in a relative short time period since all species are easily recognizable and detectable. The impact of nature management on species composition of certain biotopes can be evaluated by monitoring not only presence/absence of the multi-species group, but by additionally incorporating relative abundances of the different species; increasing abundances [compared to control situations] of species that indicate a divergence from a presupposed target can be used to alter the actual nature management scheme. A further extension of the multi-species approach is that it allows for the prioritization of site selection in a focal region: counting the number of species from a multi-species group for a certain biotope type per relevant mapping unit [e.g., km^2, the smallest grid unit used in mapping schemes in Flanders] rapidly indicates the most important areas for that specific biotope [see also **Chapter 5** were species richness from taxonomically different groups was used to determine priority conservation areas in Flanders]. Interpreting absences [or low abundances] of certain species from the multi-species group can instigate appropriate management measures or acquisition policies for surrounding sites to fulfil the needs of missing habitat specialists [this approach is comparable to the expected species-pool approach of Speight & Castella 2001]. Land development and/or nature restoration projects are other areas in which multi-species approaches can be applied; an example of a multi-species use in a land development project in Flanders can be given for the Glanville Fritillary *Melitaea cinxia* in the valley of the Grote Nete [Mol-Balen]; here, the joint presence of three other butterfly species [the Small Copper *Lycaena phlaeas*, the Common Blue *Polyommatus icarus* and the Small Heath *Coenonympha pamphilus*] appeared to be a good indicator of habitat patch suitability for the Glanville Fritillary *Melitaea cinxia* permitting the selection of local reintroduction sites for this threatened butterfly [Wallis de Vries 2001b]. When multi-species approaches are applied on large regions [e.g., the whole Flemish territory], a differentiation of multi-species lists among ecological regions [De Blust 2001] is recommended because species composition can differ among regions. On the other hand, species can find similar habitat conditions in different biotope types in the different ecological regions [e.g., the Grayling *Hipparchia semele* and the Blue-winged Grasshopper *Oedipoda caerulescens* are restricted to marram and grey dunes in the Coastal region while they occur on dry heathland in the Campine region; Decleer *et al.* 2000; Maes & Van Dyck 1999].

Multi-species groups can be compiled based on existing information but their effectiveness as 'conservation umbrella' should consequently be tested and monitored in order to evaluate and adjust these groups if necessary [**Chapter 8**]. This is possible through an integrated monitoring scheme where species, biotope types and abiotic variables are simultaneously monitored on a long-term basis [Balmford *et al.* 2003; Demeulenaere *et al.* 2002; Mc Geoch 1998]. The development of an integrated monitoring scheme is one of the most urgent tasks for nature conservation in Flanders because it will allow a more evidence-based evaluation [e.g., management, policy] and planning [e.g., site selection and configuration]. Suggestions for the development of integrated monitoring schemes in Flanders are given by Antrop *et al.* [2000] and by Demeulenaere *et al.* [2002]. Antrop *et al.* [2000] propose a monitoring scheme [outside conservation areas] in which 30 spatially stratified sampling points [squares of 1x1 km], distributed over the different traditional landscape types of Flanders, are monitored for a number of variables and species every five years [e.g., land use, desiccation, eutrophication, acidification, fragmentation, agriculture and recreation on the one hand and vascular plants, birds, amphibians and butterflies on the other]. Such an integrated monitoring scheme allows for a comparison of changes in [a selection of] biodiversity with that of land use, the environment and human activities. Ideally, the number of sampling points should be increased in the future to encompass a greater variety of landscapes and species. Demeulenaere *et al.* [2002] propose a hierarchically structured monitoring scheme in which a small number of nature reserves in Flanders [ca. 10] are monitored intensively [i.e., a large number of variables and species are monitored yearly] and a large number of nature reserves [ca. 270] is monitored with a low-intensity [i.e., a limited number of species and variables are monitored in larger time intervals]. In order to keep such integrated monitoring schemes feasible on large scales, the philosophy of multi-species approaches, as proposed in **Chapter 8**, is recommended [Van Dyck *et al.* 2001] However, this implies a significant investment in both field workers [gathering the data] and in scientists [compiling and updating the multi-species lists for the different purposes and analysing correlations between observed changes in abiotic and biotic data]. This can only be achieved through a division of tasks between volunteer organisations, policy makers and scientists. The recruitment of a large team of field workers for the gathering of data by the scientific units [e.g., the Institute of Nature

Conservation and the Institute for Forestry and Game Management] of the Flemish community would be the ideal, but given the large number of necessary sampling points, an unrealistic scenario. A closer collaboration with, and a more structural support of, volunteer organisations by the authorities is therefore more appropriate. Public scientific institutions should offer the necessary and uniform formats and protocols for the collection of the data [e.g., via an interactive website], while volunteer organisations should be supported financially to instruct local collaborators.

Communication and education in nature conservation

Practitioners and scientists usually have contrasting questions and needs [Fig. 9.2]. Bridging this gap between scientists and practitioners is of major importance if nature conservation is to become more evidence-based [Stinchcombe *et al*. 2002]. This can only be achieved through good collaboration and communication between the main actors in nature conservation: nature managers, policy makers and scientists [Balmford *et al*. 2003; Jacobson & Robinson 1990; Jacobson & McDuff 1998a; Robertson & Hull 2001; Soulé 1986; van Leeuwen & de Ridder 1998]. This task of communication [translating results of scientific research into practical guidelines and, vice versa, converting practical field questions into proposals for scientific research] is best performed by scientifically trained staff with a certain amount of experience both in conservation practice and in scientific research and with regular contacts with practitioners [Pullin & Knight 2003; Salafsky *et al*. 2002]. This communication should be promoted by both the NGO's and by the responsible scientific units of the Flemish authorities.

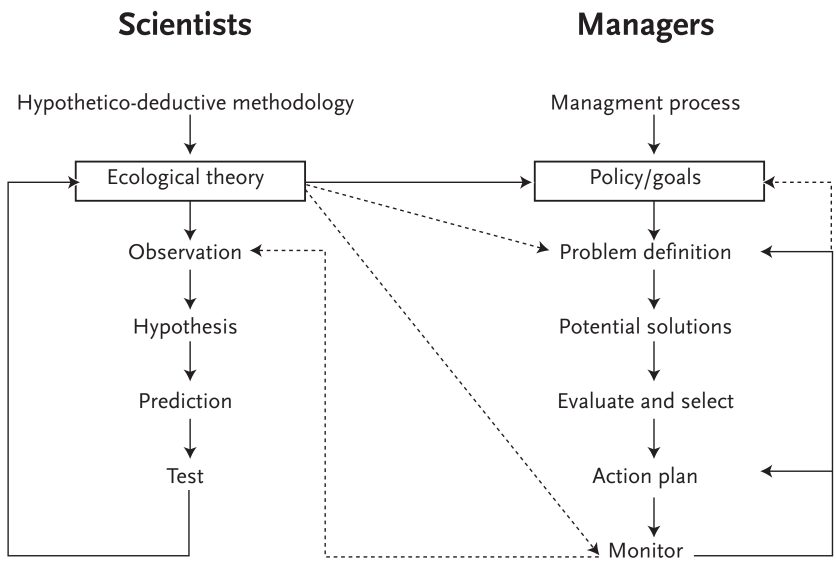

Figure 9.2. Contrasting positions of the practitioner and the scientist [Pullin 2002a].

Although non-governmental conservation organisations in Flanders [e.g. Natuurpunt, Stichting Limburgs Landschap] and the authorities allow and even welcome scientific studies in their nature reserves [as 'open-air laboratories'; Walters 1986; see **Chapters 6-8** for examples], scientific research in reserves is usually done on an ad-hoc basis and communication between practitioners and scientists is often lacking [Gerber & Schultz 2001; Hecht & Parkin 2001; Meffe *et al*. 1998; Pullin & Knight 2003; Salafsky *et al*. 2002]. Wardens or local field workers are, therefore, often not aware of the progress or the results of such research. The gap between scientists and practitioners would certainly become more narrow through the regular organization of integrative workshops where scientists inform managers about new insights in nature management techniques and impacts or policy makers about examples of the use of species information in site selections, land development, etc. [Cort 1996; Meffe *et al*. 1998; Prendergast *et* *al*. 1999; Pullin & Knight 2003]. Such a forum can, additionally, lead to interesting scientific research in conservation biology originating from practitioners needs in the field [Pullin & Knight 2003]. Not all scientific research, however, is translatable into practical guidelines: many, more fundamental scientific questions are often studied in hypothetical situations with assumptions that are unrealistic in the field. On the other hand, some of the conservation biology research, although relevant from a scientific point of view, does not provide answers to the problems conservation practitioners are facing in the field.

In order to be usable for practitioners, information on the distribution and scientific research on the ecology of species [e.g. habitat use, mobility, minimum patch areas, etc.] needs to be published in reference works accessible to a large public [Robertson & Hull 2001; van Leeuwen & de Ridder 1998]. Care should be taken, however, not to confound the vulgarizing nature of such reference works with the scientific methods applied to compile them. In order to generate new information and new research continuously, it is of extreme importance towards both practitioners and scientists to make a clear distinction between what is currently known and what is assumed. Research results that are only published in scientific journals are usually inaccessible to practitioners and will therefore not be implemented. If such research was intended to give management advice to practitioners, a practical and accessible publication with detailed descriptions should also be made available. An example of the latter is given in the species action plan for the Alcon Blue butterfly *Maculinea alcon* [Vanreusel *et al*. 2000]. Species action plans or management plans for nature reserves are a good opportunity to bridge the gap between scientists and practitioners [Pullin 2002a; Stinchcombe *et al*. 2002]. Most management plans are compiled by the local wardens and describe the site and, if available, the actual and historical species richness together with management aims and methods.

Only rarely, a monitoring scheme for management evaluation is added. But, most of the proposed actions are not underpinned by scientific evidence and largely remain experience-based [Pullin 2002a]. A critical screening of management and species action plans by scientists could incite practitioners to a more evidence-based approach for suggesting conservation actions [Clark *et al*. 2002; Meffe *et al*. 1998].

In addition to a centralised distribution and ecological data base, a publically accessible data base with nature conservation research projects could make possible end users aware of existing research results. The fact that many of these projects are funded by different 'organisations' makes it even more important to have an overview of [the results of] all past and current research subjects [e.g., Federale Diensten voor Wetenschappelijke, Technische en Culturele aangelegenheden [Federal Office for Scientific, Technical and Cultural Affairs] - DWTC, Milieu- en Natuur-fonds [Environment and Nature funding of the Flemish government] - MINA, Toegepast Wetenschappelijk Onderzoek inzake Leefmilieu [Applied Scientific Research concerning the Environment] - TWOL, het Instituut voor de Aanmoediging van Innovatie door Wetenschap en Technologie in Vlaanderen [Institute for the Encouragement of Innovation through Science and Technology in Flanders] - IWT, het Fonds voor Wetenschappelijk Onderzoek [Fund for Scientific Research Flanders-Belgium] - FWO, etc.]. Both practitioners [management] and scientists [further research] can make use of the results of these projects. Reports of the different research projects should be made readily available to all possible end users. Furthermore, in order to communicate effectively, scientist should know about the form and the timing of the information needed by practitioners and by policy makers [Theobald *et al*. 2000]. Therefore, regular consultations with all nature conservation actors should be organized in order to adapt the existing communication channels or even scientific research schemes.

Scientists are currently poorly trained to communicate the results of their research to field workers or to policy makers [Jacobson & McDuff 1998b]. Incorporating packages on communication techniques in the present courses of biology or agronomy students could rectify this deficiency [Jacobson & McDuff 1998a,b]. Nature conservation would benefit greatly from the organisation of specialization courses in 'translating' scientific research [published in international specialist literature] into practical conservation guidelines in nature management, policy making, restoration projects, land development, etc. Apart from informing people in the field about existing information, education about the possibilities of more evidence-based approaches in general and on the use of species in particular is an equally important, but a neglected field in nature conservation. Most courses about species [but also mapping or monitoring schemes] are strictly taxonomically based. A more practical approach could be to base courses on concrete questions of practitioners [e.g., how do I judge whether the management scheme I am applying in a certain biotope type is effective?] or policy makers [e.g., on what basis should we decide to incorporate sites into the Flemish Ecological Network?]. Such approaches can make use of information on a large number of species [multi-species groups crossing taxonomic boundaries] and are more effective for planning and evaluating nature conservation actions than pure taxonomic knowledge [**Chapters 6-8**].

Towards scientific methods and standards for the use of species in conservation practice

Fig. 9.3 gives a schematic overview of how the different actors and information sources in nature conservation could interact with one another. Both the severity of the actual biodiversity crisis at a global and particularly at a Flemish level and the more general constraint of limited budget resources, demand an optimal use of existing [species] information and a maximization – including a better, reciprocal tuning – of the efforts made by the three main actors in nature conservation, i.e., practitioners, scientists and policy makers.

In order to encourage the use of more scientific methods and standards in conservation practice, Pullin & Knight [2001] proposed a series of steps to move nature conservation towards a more evidence-based action. One of the central aspects of their approach is that of the so-called systematic review, a process whereby the quality and relevance of published [or unpublished] research is judged and translated into a usable format. The government's conservation research institution, the Institute of Nature Conservation, is ideally placed to play a prominent role towards a more evidence-based nature conservation in Flanders. To achieve this objective it can:

1 prepare a proposal for a policy on evidence-based action and play a key role for its implementation;
2 identify priority research areas [e.g., management aims and tools, site selection ...] for systematic review with a proper allocation of personnel and structural fundings;
3 instigate research in areas where information is found lacking;
4 set mechanisms and standards of conservation practice for public nature reserves and other conservation related issues of other statutory bodies in consultation with the 'scientific administrators' of the Nature [Afdeling Natuur], the Forest and Parks Departments [Afdeling Bos en Groen], the Flemish Land Agency [VLM], the provinces ...

Aditionally, a critical screening of management plans of private nature reserves [many of them largely subsidized with public money] by the Institute of Nature Conservation followed by a constructive feedback towards practitioners should incite the different NGO's [e.g., Natuurpunt, Stichting Limburgs Landschap ...] to apply a more evidence-based approach in conservation actions as well [Clark *et al*. 2002; Meffe *et al*. 1998; Pullin & Knight 2003]. Such an approach, co-ordinated by the Institute of Nature Conservation, gives an added value to the actions undertaken by the NGO's, that emerges from bringing monitoring data together from both NGO practitioners and scientists [at the Institute of Nature Conservation and at the Nature and Forest and Parks Departments] in order to develop scientifically sound long-term time series. Preferably, projects for mapping schemes and species action plans, both from NGO's and from statutory bodies, should also be screened rigorously on their scientific content in order to ensure an optimal use of the obtained information in field actions. On the other hand, scientists [working at universities, statutory bodies or NGO's] and practitioners should be able to make their information readily available in a usable format in order to permit decision-makers [e.g., the scientific administrators of the Nature or the Forest and Parks Departments] to properly evaluate and choose the best conservation options. Presently, the Institute of Nature Conservation is constructing, in collaboration with the NGO's, the Nature and Forest and Park Departments, the Flemish Land Agency and the provinces, an internet-based interface where essential species information [e.g., distribution, ecology, threat status ...] and site information [e.g., biotope descriptions, management, reserve status ...] can be easily exchanged between the different actors in the field of nature conservation [Fig. 9.3]. Formalising collaborations between policy makers, scientists, practitioners and volunteers makes it more likely that existing information is provided and used, but also that knowledge gaps in nature conservation will be detected more rapidly [Pullin & Knight 2003].

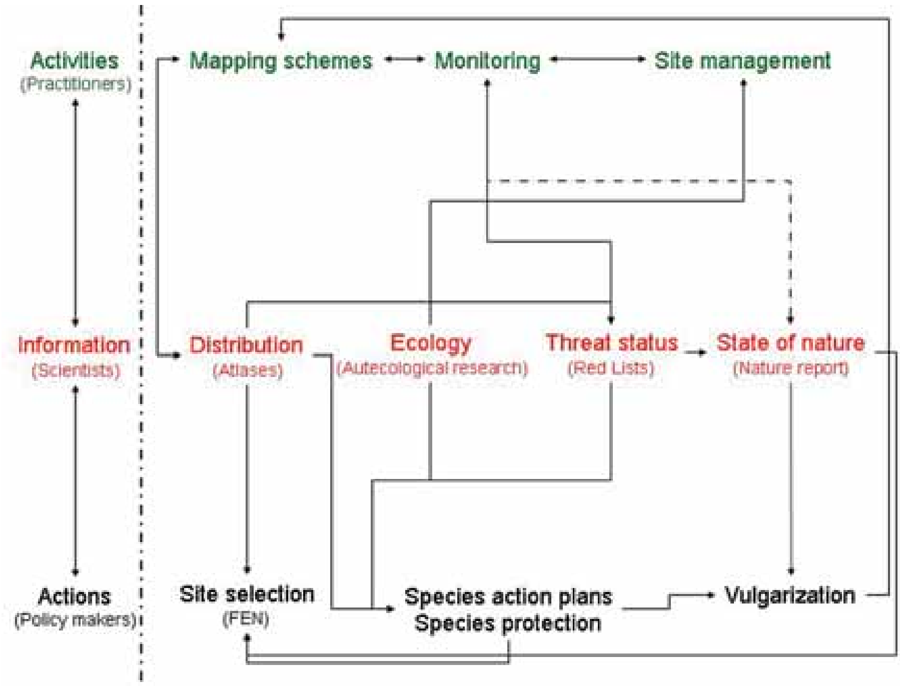

Figure 9.3. Interactions between the different actors and information levels in nature conservation in Flanders.

The use of indicator species in nature management and policy making.

The case of invertebrates in Flanders [northern Belgium]

In Flanders, as in most other NW-European countries, decisions in nature conservation are often non-ecologically based. In the best case, such decisions are based on the presence of certain biotope types [i.e., site-based] or on the maintenance of ecological processes. Species-specific information is, up-to-date, only rarely used in policy making or in evaluating or planning site selection or management. There is, however, a growing interest in using [indicator] species as tools or as goals in nature conservation in Flanders. The use of short-cut concepts like indicators is very appealing, but at the same time problematic because their effectiveness is usually assumed but rarely tested. Furthermore, a single indicator species is unlikely to encompass the ecological requirements of a large number of sympatric species or all characteristic habitat features in a certain biotope. Therefore, the use of multispecies approaches has been promoted for different issues in nature conservation. Invertebrates constitute 75% of all biodiversity, but are often ignored as possible tools or goals in nature conservation. However, the fact that many invertebrates occupy narrow niches, use biotopes on a small scale, have a low mobility and react rapidly to changes in the environment, makes their 'information content' complementary to that of other better known species such as birds, mammals or plants.

In the different chapters of this thesis, the extent to which the use of [multi-] species information provides a surplus value to nature conservation and policy making in Flanders is explored. The focus is on invertebrates and, here, butterflies are often used as model organisms. **Chapter 2** describes the application of a uniform and quantitative Red List methodology and the use of internationally accepted Red List categories that are now widely used in Flanders. This facilitates both the comparison of threat statuses of species from different taxonomic groups as well as the communication with international conservation bodies [e.g., IUCN, EU, Council of Europe].

In **Chapter 3**, ecological and distributional species information, in this case of butterflies, is linked to changes in land use during the 20th century. This analysis revealed that habitat loss, fragmentation and eutrophication are the main causes of the strong decline of butterflies in Flanders. Butterflies can thus serve as sensitive 'indicators' for the assessment of the state of nature in Flanders. Mapping schemes often have to deal with severe biases in both time and space. Bias in time is caused by the fact that recent records are usually more numerous and more accurate than historical ones [where often only the name of a city is available]; bias in space is caused by the uneven geographic distribution of recorders.

To deal with such problems, modelling techniques [**Chapter 4**] allow for the incorporation of ill-surveyed regions in conservancy policies [e.g., indicating potentially species-rich zones]. Furthermore, modelling techniques can optimise mapping schemes [by indicating volunteers what regions are potentially species-rich]. Founding the delineation of areas for conservation on a single taxonomic group or species is usually not appropriate because the assumption that species richness coincides among different species groups, has proven to be false.

Applying a multispecies approach to determine potentially species-rich areas in Flanders is explored in **Chapter 5**. To overcome biases in mapping intensity and in geographical extent, we first apply modelling techniques to five well-investigated taxonomic groups [plants, dragonflies, butterflies, herpetofauna and birds] to predict the distribution of species richness separately. Within Flanders, the four faunal groups appeared to be relatively good indicators for each other, but species richness distribution in plants did not coincide well with that of the four faunal groups.

Detailed autecological research on invertebrates in threatened biotopes [wet heathland in our case] indicates that invertebrates can add useful information to the primarily site-based nature conservation in Flanders. **Chapter 6** focuses on the composition of ants in wet heathland and how this information can be incorporated into management schemes. Adapting management schemes with this knowledge can seriously increase the number of nesting sites for the dominating ant species in this threatened biotope. This can locally be beneficial for the Alcon Blue butterfly *Maculinea alcon*, one of the myrmecophilous species that has become very rare on wet heathlands in Flanders.

In **Chapter 7**, we delimited so-called functional units for the conservation of this European-wide threatened butterfly. We make use of detailed knowledge on ecology, mobility, distribution and colonization ability to delimit three types of units within which different management intensities should be implemented. The use of clearly defined conservation units and the proposition of detailed management measures for the conservation of a threatened species, greatly facilitates communication with practitioners.

Finally, since single species conservation does not necessarily ensure the conservation of other sympatric species, we also apply a multi-species approach for wet heathlands in Flanders [**Chapter 8**]. Here, the use of a set of easily recognisable and easily detectable species from different taxonomic groups [2 birds, 2 plants, 2 dragonflies, 2 butterflies and 1 grasshopper] appeared to be a better 'conservation umbrella' for wet heathlands than the single use of the Alcon Blue butterfly *Maculinea alcon*.

In a final chapter, the use of [indicator] species in nature conservation in Flanders is discussed [**Chapter 9**]. Here, methods are given for a better use of the available information [e.g., in mapping schemes, Red Lists and species action plans] in nature conservation. Guidelines are also given for the gathering of relevant information that is presently lacking for an adequate use of species. Evidence-based approaches [contrary to the actually more experience-based ones] and the use of a larger suite of indicator species for a wide variety of nature conservation purposes [i.e., the description of biotope types, habitat quality evaluation, assessing the impact of nature management, nature restoration or land development projects, the selection of sites for the Flemish Ecological Network, etc.] are advocated. Finally, the need for a better communication among the different actors in nature conservation [scientists, policy makers and practitioners] is emphasized.

Utilisation d'espèces indicatrices dans la gestion et la politique de la nature. Le cas des invertébrés en Flandre

En Flandre, comme dans la plupart des autres pays du nord-ouest de l'Europe, les décisions prises en matière de conservation de la nature sont souvent basées sur des principes non-écologiques. Dans le meilleur des cas, de telles décisions sont basées sur la présence de certains types d'habitat [c'est-à-dire basées sur le site] ou sur la préservation de processus écologiques. Les informations spécifiques aux espèces ne sont, jusqu'à présent, que rarement utilisées dans l'établissement des politiques de conservation ou dans l'évaluation ou la planification de la sélection et de la gestion des sites. Cependant, l'utilisation d'espèces [indicatrices] comme outils ou objectifs en conservation de la nature en Flandre semble susciter un intérêt croissant. L'utilisation de 'concepts raccourcis' tels que des indicateurs biologiques est à la fois très attirante et problématique puisque son efficacité est généralement supposée mais rarement testée. De plus, il est peu probable qu'une seule espèce indicatrice puisse englober les exigences écologiques d'un grand nombre d'espèces sympatriques ou toutes les caractéristiques typiques d'habitats dans un certain biotope. Dès lors, l'utilisation d'approches multi-spécifiques a été promue pour différentes problématiques en conservation de la nature. Les invertébrés représentent 75% de toute la biodiversité, mais sont souvent ignorés comme étant des outils ou des objectifs possibles en conservation de la nature. Cependant, le fait que beaucoup d'invertébrés occupent des niches restreintes, utilisent des biotopes sur une petite échelle, ont une faible mobilité et réagissent rapidement aux changements de l'environnement, rend leur 'pouvoir informatif' complémentaire à celui d'autres espèces mieux connues telles que les oiseaux, les mammifères ou les plantes.

Dans les différents chapitres de cette thèse, l'ampleur avec laquelle l'utilisation d'informations [multi-]spécifiques ajoute une valeur supplémentaire à la conservation de la nature et dans l'établissement de politiques en Flandre est explorée. Nous nous concentrons sur les invertébrés et, dans cette étude, les papillons sont souvent utilisés comme modèles. Le **Chapitre 2** décrit l'application d'une méthodologie uniforme et quantitative de liste rouge et l'utilisation de catégories de liste rouge maintenant acceptées internationalement et largement utilisées en Flandre. Ceci facilite à la fois la comparaison des statuts de menace d'espèces de différents groupes taxonomiques et la communication avec les centres internationaux de conservation [par exemple, IUCN, EU, Conseil de l'Europe].

Dans le **Chapitre 3**, les informations relatives à l'écologie et la distribution des espèces, en l'occurrence des papillons, est mise en relation avec les changements d'utilisation du territoire durant le $20^{ème}$ siècle. Cette analyse révèle que la perte d'habitat, la fragmentation et l'eutrophisation sont les causes principales du sévère déclin des papillons en Flandre. Les papillons peuvent donc servir d'indicateurs sensibles pour l'évaluation de l'état de la nature en Flandre. Les projets cartographiques doivent souvent faire face à des biais sévères à la fois dans le temps et dans l'espace. Les biais temporels sont dus au fait que les données récentes sont habituellement plus nombreuses et plus précises que les données historiques [où souvent seul le nom d'une ville est disponible]. Les biais spatiaux sont eux dus à la distribution géographique irrégulière des observateurs.

Pour gérer de tels problèmes, des techniques de modélisation [**Chapitre 4**] permettent l'incorporation de régions sous-prospectées dans des stratégies de conservation [par exemple en indiquant des zones potentiellement riches en espèces]. De plus, les techniques de modélisation peuvent optimiser les projets cartographiques [en indiquant aux volontaires quelles sont les régions potentiellement plus riches]. Baser la délimitation des zones à conserver sur une seule espèce ou sur un groupe taxonomique n'est généralement pas approprié car l'hypothèse selon laquelle la richesse spécifique coïncide au sein de différents groupes s'est révélée fausse.

L'application d'une approche multi-spécifique pour déterminer les zones potentiellement riches en espèces en Flandre est explorée dans la **Chapitre 5**. Pour surmonter les biais dus à l'intensité de l'échantillonnage et dans la couverture géographique, nous appliquons d'abord des techniques de modélisation sur 5 groupes taxonomiques bien inventoriés [plantes, libellules, papillons, herpétofaune et oiseaux] pour prédire séparément la distribution de la richesse spécifique. En Flandre, les quatre groupes fauniques apparaissent comme étant d'assez bons indicateurs les uns des autres, mais la distribution de la richesse spécifique chez les plantes ne coïncide pas bien avec celle des quatre autres groupes fauniques. Des recherches détaillées sur l'autécologie chez les invertébrés dans des biotopes menacés [landes humides dans notre cas] indiquent que les invertébrés peuvent apporter une information utile à la conservation de la nature, principalement basée, en Flandre, sur les sites.

Le **Chapitre 6** se penche sur les communautés de fourmis dans les landes humides et sur la façon d'intégrer cette information dans des stratégies de gestion. Adapter de la sorte les stratégies de gestion peut augmenter sérieusement le nombre de sites de nidification de l'espèce dominante de fourmi dans ce milieu menacé. Cela peut être localement bénéfique pour le papillon *Maculinea alcon* [le Protée], une des espèces myrmécophiles devenue très rare dans les landes humides en Flandre.

Dans le **Chapitre 7**, nous avons délimité des unités fonctionnelles pour la conservation de ce papillon menacé à l'échelle européenne. Nous utilisons nos connaissances détaillées sur son écologie, sa mobilité, sa distribution et sa capacité de colonisation afin de délimiter trois types d'unités dans les-quelles différentes intensités de gestion devraient être appliquées. L'utilisation d'unités de conservation clairement définies et la proposition de mesures de gestion détaillées pour la conservation d'une espèce menacée facilite grandement la communication avec les gestionnaires.

Finalement, comme la conservation d'une seule espèce n'assure pas nécessairement la conservation d'autres espèces sympatriques, nous appliquons aussi une approche multi-spécifique pour les landes humides en Flandre [**Chapitre 8**]. Dans cette étude, l'utilisation d'un lot d'espèces facilement identifiables et détectables appartenant à différents groupes taxonomiques [2 oiseaux, 2 plantes, 2 libellules, 2 papillons et 1 sauterelle] est apparue comme un meilleur «parapluie de conservation» pour les landes humides que l'utilisation d'une espèce unique telle que *Maculinea alcon*.

Dans le chapitre final, l'utilisation d'espèces [indicatrices] en conservation de la nature en Flandre est discutée [**Chapitre 9**]. Des méthodes sont proposées pour une meilleure utilisation de l'information disponible [par exemple dans les projets cartographiques, les listes rouges et les plans de protection d'espèces] en conservation de la nature. Des lignes directrices sont aussi données pour rassembler les informations relevantes qui manquent actuellement pour une utilisation adéquate des espèces. Les approches basées sur des preuves [contrairement aux approches actuelles plus basées sur l'expérience] et l'utilisation d'une plus grande série d'espèces indicatrices pour une grande variété d'objectifs de conservation [c'est-à-dire la description du type d'habitat, l'évaluation de la qualité de l'habitat, l'estimation de l'impact de la gestion, la restauration de la nature ou les projets de développement d'aménagement du territoire, la sélection de sites pour le réseau écologique flamand, etc.] sont défendues. Enfin, la nécessité d'une meilleure communication entre les différents acteurs de la conservation de la nature [scientifiques, décideurs politiques et gestionnaires] est mise en évidence.

Het gebruik van indicatorsoorten in het natuurbeheer en -beleid. Ongewervelden in Vlaanderen als voorbeeld

In Vlaanderen, net zoals in de meeste andere NW-Europese landen, zijn natuurbehoudsbeslissingen vaak niet gebaseerd op ecologische argumenten. In het beste geval worden zulke beslissingen genomen op basis van de aanwezigheid van bepaalde biotopen [d.i., gebiedsgericht] of op basis van het behoud van ecologische processen. Soort-specifieke informatie is, tot op heden, slechts zelden gebruikt bij het evalueren of selecteren van gebieden of in natuurbeleidsdomeinen. Er is echter een toenemende interesse voor het gebruik van soorten als instrumenten of als doelen in het natuurbehoud in Vlaanderen. Het gebruik van short-cut concepten zoals indicatorsoorten is bijzonder aantrekkelijk, maar tegelijkertijd problematisch omdat hun doeltreffendheid vaak enkel verondersteld wordt, maar zelden getest. Bovendien is een enkele indicatorsoort zelden in staat om alle ecologische behoeften van een groot aantal andere soorten of een groot deel van de abiotische biotoopkarakteristieken te omvatten. Daarom werd recent het gebruik van een multi-soortenaanpak in verschillende natuurbehoudstoepassingen voorgesteld. Ongewervelden vormen 75% van alle biodiversiteit, maar worden vaak over het hoofd gezien als mogelijke instrumenten of doelen in het natuurbehoud. Het feit dat vele ongewervelden echter een smalle niche innemen, biotopen op een kleine schaal gebruiken, weinig mobiel zijn en snel reageren op veranderingen in hun omgeving, maakt hun 'informatie-inhoud' complementair aan die van beter gekende soorten zoals vogels, zoogdieren of planten.

In de verschillende hoofdstukken van deze thesis wordt nagegaan in welke mate het gebruik van [multi-]soorten informatie een meerwaarde kan zijn voor het natuurbehoud en –beleid in Vlaanderen. De aandacht gaat daarbij vooral naar ongewervelden in het algemeen en naar dagvlinders in het bijzonder, die hier als modelorganismen gebruikt worden.

Hoofdstuk 2 beschrijft het toepassen van een uniforme en kwantitatieve Rode-Lijstmethodiek en het gebruik van internationaal aanvaarde Rode-Lijstcategorieën, die nu algemeen gebruikt worden in Vlaanderen. Deze uniformiteit vergemakkelijkt zowel het vergelijken van de bedreigingsgraad van soorten uit verschillende taxonomische groepen als de communicatie met internationale instanties [bv. IUCN, EU, Raad van Europe].

In **Hoofdstuk 3** wordt ecologische en verspreidingsinformatie van soorten, in dit geval dagvlinders, gekoppeld aan veranderingen in landgebruik gedurende de 20ste eeuw. Deze analyse toonde aan dat verlies van geschikt habitat, habitatfragmentatie en vermesting de voornaamste oorzaken waren van de achteruitgang van dagvlinders in Vlaanderen. Dagvlinders kunnen op die manier mede gebruikt worden als gevoelige indicatoren voor het beschrijven van de toestand van de natuur. Inventarisatieprojecten hebben vaak te maken met ongelijke speiding van de gegevens zowel in de tijd [er zijn vaak veel meer recente dan historische gegevens beschikbaar] als in de ruimte [door de ongelijke geografische verdeling van waarnemers over Vlaanderen].

Om zulke problemen gedeeltelijk op te vangen kunnen modelleertechnieken [voor het aanduiden van potentieel soortenrijke gebieden] gebruikt worden waardoor onder- of helemaal niet-geïnventariseerde regio's betrokken kunnen worden bij natuurbehoudsvragen [**Hoofdstuk 4**]. Bovendien kunnen dergelijke technieken inventarisatieprojecten merkelijk optimaliseren door waarnemers aan te geven waar potentieel soortenrijke regio's gelegen zijn. Het baseren van het selecteren van gebieden op een enkele soort of taxonomische groep is meestal niet aangewezen aangezien de veronderstelling dat soortenrijkdom tussen verschillende taxonomische groepen gecorreleerd is, fout blijkt.

Het toepassen van een multi-soortenaanpak voor het afbakenen van potentieel soortenrijke gebieden in Vlaanderen, wordt onderzocht in **Hoofdstuk 5**. Om de ongelijkheid in inventarisatie-inspanning zowel in tijd als in ruimte te compenseren, passen we eerst modelleertechnieken toe op vijf goedgeïnventariseerde taxonomische groepen [planten, libellen, dagvlinders, amfibieën en reptielen en vogels] om de soortenrijkdom per groep te voorspellen. In Vlaanderen blijken de vier faunagroepen relatief goede indicatoren voor elkaars soortenrijkdom te zijn, maar was de verspreiding van de plantensoortenrijkdom veel minder goed gecorreleerd met die van de faunagroepen. Gedetailleerd autecologisch onderzoek naar invertebraten in bedreigde biotopen [zoals natte heide] toont aan dat ongewervelden nuttige informatie kunnen toevoegen aan de voornamelijk gebiedsgerichte aanpak in Vlaanderen.

Hoofdstuk 6 gaat dieper in op de samenstelling van de mierenfauna op natte heide en op hoe deze informatie gebruikt kan worden bij het opstellen of aanpassen van beheersplannen. Het aanpassen van het beheer met behulp van deze kennis, kan het aantal potentiële nestplaatsen van de typische mierenfauna aanzienlijk verhogen. Deze maatregel kan lokaal zeker ten goede komen van het Gentiaanblauwtje *Maculinea alcon*, een zeldzame myrmecofiele soort van natte heide.

In **Hoofdstuk 7** bakenen we functionele behoudseenheden af voor deze, ook op Europese schaal, bedreigde dagvlinder. We maken hiervoor gebruik van gedetailleerde kennis over de ecologie, verspreiding, mobiliteit en kolonisatie-capaciteit om drie types behoudseenheden af te bakenen waarin natuurbeheer met verschillende intensiteiten uitgevoerd moet worden. Het gebruik van duidelijk afgebakende behoudseenheden en het voorstellen van gedetailleerd natuurbeheersvoorstellen voor het behoud van een bedreigde soort vergemakkelijkt de communicatie met mensen in het veld aanzienlijk.

Aangezien het behoud van een enkele soort zelden het behoud van een hele reeks samenlevende soorten kan verzekeren, passen we eveneens een multi-soortenaanpak toe voor het beheer en het behoud van natte heide in Vlaanderen [**Hoofdstuk 8**]. Het gebruik van een groep gemakkelijk herkenbare en determineerbare soorten uit verschillende taxonomische groepen [2 vogels, 2 planten, 2 libellen, 2 dagvlinders en 1 sprinkhaan] bleek een betere 'behoudsparaplu' te zijn dan het exclusief gebruik van het Gentiaanblauwtje *Maculinea alcon*.

In een laatste hoofdstuk wordt het gebruik van [indicator] soorten in het natuurbehoud in Vlaanderen bediscussieerd [**Hoofdstuk 9**]. Hier worden suggesties gegeven voor een beter gebruik van de beschikbare informatie [door een betere ontsluiting van bv. verspreidingsgegevens, Rode Lijsten en soortbeschermingsplannen] in het natuurbehoud. Tevens worden richtlijnen gegeven voor het verzamelen van relevante, maar momenteel ontbrekende, informatie voor een optimaler gebruik van soorten. Een wetenschappelijk onderbouwde aanpak [in tegenstelling met de momenteel vaker toegepaste ervaringsgerichte aanpak] en het gebruik van een groter aantal [indicator] soorten voor een grote verscheidenheid aan natuurbehoudstoepassingen [bv. biotoopbeschrijvingen, het evalueren van habitatkwaliteit, inschatten van de effecten van natuurbeheer, opvolgen van natuurontwikkelings- of landinrichtingsprojecten, het selecteren van gebieden voor het Vlaams Ecologisch Netwerk, enz.] worden bepleit. Tenslotte wordt de nood aan een betere communicatie tussen de verschillende actoren in het natuurbehoud [wetenschappers, natuurbeheerders en beleidsmensen] benadrukt.

Scientific and Dutch names of species mentioned in the text

Butterflies [Vlinders] – [Karsholt & Razowski 1996]

Aglais urticae [Kleine vos]
Anthocharis cardamines [Oranjetipje]
Apatura iris [Grote weerschijnvlinder]
Aphantopus hyperantus [Koevinkje]
Aporia crataegi [Groot geaderd witje]
Araschnia levana [Landkaartje]
Argynnis adippe [Adippevlinder]
Argynnis aglaja [Grote parelmoervlinder]
Argynnis niobe [Duinparelmoervlinder]
Argynnis paphia [Keizersmantel]
Aricia agestis [Bruin blauwtje]
Boloria euphrosyne [Zilvervlek]
Boloria selene [Zilveren maan]
Callophrys rubi [Groentje]
Carterocephalus palaemon [Bont dikkopje]
Celastrina argiolus [Boomblauwtje]
Coenonympha glycerion [Roodstreephooibeestje]
Coenonympha hero [Zilverstreephooibeestje]
Coenonympha pamphilus [Hooibeestje]
Coenonympha tullia [Veenhooibeestje]
Colias croceus [Oranje luzernevlinder]
Colias hyale [Gele luzernevlinder]
Cupido minimus [Dwergblauwtje]
Erebia ligea [Boserebia]
Erynnis tages [Bruin dikkopje]
Euphydryas aurinia [Moerasparelmoervlinder]
Gonepteryx rhamni [Citroenvlinder]
Hesperia comma [Kommavlinder]
Heteropterus morpheus [Spiegeldikkopje]
Hipparchia semele [Heivlinder]
Hipparchia statilinus [Kleine heivlinder]
Inachis io [Dagpauwoog]
Iphiclides podalirius [Koningspage]
Issoria lathonia [Kleine parelmoervlinder]
Lasiommata megera [Argusvlinder]
Leptidea sinapis [Boswitje]
Limenitis camilla [Kleine ijsvogelvlinder]
Limenitis populi [Grote ijsvogelvlinder]
Lycaena dispar [Grote vuurvlinder]
Lycaena phlaeas [Kleine vuurvlinder]
Lycaena tityrus [Bruine vuurvlinder]
Maculinea alcon [Gentiaanblauwtje]
Maculinea arion [Tijmblauwtje]
Maculinea teleius [Pimpernelblauwtje]
Maculinea nausithous [Donker pimpernelblauwtje]
Maniola jurtina [Bruin zandoogje]
Melanargia galathea [Dambordje]
Melitaea athalia [Bosparelmoervlinder]
Melitaea cinxia [Veldparelmoervlinder]
Melitaea diamina [Woudparelmoervlinder]
Neozephyrus quercus [Eikenpage]
Nymphalis antiopa [Rouwmantel]
Nymphalis polychloros [Grote vos]
Ochlodes venata [Groot dikkopje]
Papilio machaon [Koninginnenpage]
Pararge aegeria [Bont zandoogje]
Pieris brassicae [Groot koolwitje]
Pieris napi [Klein geaderd witje]
Pieris rapae [Klein koolwitje]
Plebeius argus [Heideblauwtje]
Plebeius idas [Vals heideblauwtje]
Polygonia c-album [Gehakkelde aurelia]

Polyommatus icarus [Icarusblauwtje]
Polyommatus semiargus [Klaverblauwtje]
Pyrgus armoricanus [Bretons spikkeldikkopje]
Pyrgus malvae [Aardbeivlinder]
Pyronia tithonus [Oranje zandoogje]
Satyrium ilicis [Bruine eikenpage]
Satyrium w-album [Iepenpage]
Thecla betulae [Sleedoornpage]
Thymelicus lineola [Zwartsprietdikkopje]
Thymelicus sylvestris [Geelsprietdikkopje]
Vanessa atalanta [Atalanta]
Vanessa cardui [Distelvlinder]

Ants [**Mieren**] – [Boer *et al.* 2003]

Anergates atratulus [Woekermier]
Formica cunicularia [Bruine baardmier]
Formica fusca [Grauwzwarte mier]
Formica lusatica [Duinbaardmier]
Formica polyctena [Kale bosmier]
Formica pratensis [Zwartrugbosmier]
Formica rufa [Behaarde bosmier]
Formica rufa x polyctena [Formica rufa-complex]
Formica rufibarbis [Rode baardmier]
Formica sanguinea [Bloedrode roofmier]
Formica transkaucasica [Veenmier]
Formicoxenus nitidulus [Glanzende gastmier]
Lasius brunneus [Boommier]
Lasius emarginatus [Muurmier]
Lasius flavus [Gele weidemier]
Lasius fuliginosus [Glanzende houtmier]
Lasius jensi [Puntschubmier]
Lasius meridionalis [Veldmier]
Lasius mixtus [Wintermier]
Lasius myops [Kleinoogweidemier]
Lasius niger [Wegmier]
Lasius platythorax [Humusmier]
Lasius psammophilus [Buntgrasmier]
Lasius sabularum [Breedschubmier]
Lasius umbratus [Schaduwmier]
Leptothorax acervorum [Behaarde slankmier]
Leptothorax affinis [Boomslankmier]
Leptothorax muscorum [Mosslankmier]
Leptothorax nylanderi [Bosslankmier]
Myrmecina graminicola [Oprolmier]
Myrmica lonae [Lepelsteekmier]
Myrmica microrubra [Gaststeekmier]
Myrmica rubra [Gewone steekmier]
Myrmica ruginodis [Bossteekmier]
Myrmica rugulosa [Kleine steekmier]
Myrmica sabuleti [Zandsteekmier]
Myrmica scabrinodis [Moerassteekmier]
Myrmica schencki [Kokersteekmier]
Myrmica specioides [Duinsteekmier]
Myrmica sulcinodis [Heidesteekmier]
Polyergus rufescens [Amazonemier]
Ponera coarctata [Gewone staafmier]
Solenopsis fugax [Diefmier]
Stenamma debile [Gewone drentelmier]
Stenamma westwoodi [Engelse drentelmier]
Strongylognathus testaceus [Sabelmier]
Tapinoma ambiguum [Heidedraaigatje]
Tapinoma erraticum [Mergellanddraaigatje]
Tetramorium caespitum [Zwarte zaadmier]
Tetramorium impurum [Bruine zaadmier]

Other invertebrates [Andere ongewervelden]

Araeoncus crassiceps [Arrogant voorkopje]
Ceriagrion tenellum [Koraaljuffer]
Leucorrhinia dubia [Venwitsnuitlibel]
Metrioptera brachyptera [Heidesabelsprinkhaan]
Metrioptera roeseli [Greppelsprinkhaan]
Numenius arquata [Wulp]
Oedipoda caerulescens [Blauwvleugelsprinkhaan]
Saxicola torquata [Roodborsttapuit]
Somatochlora arctica [Hoogveenglanslibel]

Vascular plants [Hogere planten] – [Biesbrouck *et al*. 2001]

Calluna vulgaris [Struikhei]
Deschampsia flexuosa [Bochtige smele]
Erica tetralix [Gewone dophei]
Eriophorum angustifolium [Veenpluis]
Gentiana pneumonanthe [Klokjesgentiaan]
Molinia caerulea [Pijpenstrootje]
Narthecium ossifragum [Beenbreek]
Rhynchospora sp. [Snavelbies sp.]
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Ministerie van de
Vlaamse Gemeenschap

Instituut voor Natuurbehoud

Colofon

D/2004/3241/103
ISBN 90-403-0204-9
NUR 922

Auteur
Dirk Maes
Verantwoordelijke uitgever
Eckhart Kuijken [Algemeen directeur]
Vormgeving
Mariko Linssen
Druk
Drukkerij van de Vlaamse Gemeenschap
Adres
Instituut voor Natuurbehoud, Kliniekstraat 25, 1070 Brussel
T: 02 558 18 11 F: 02 558 18 05
www.instnat.be info@instnat.be

Wijze van citeren / how to cite

Maes D. [2004]. The use of indicator species in nature management and policy making. The case of invertebrates in Flanders [northern Belgium]. Institute of Nature Conservation, Brussels. 291 pp.

Maes D. & Van Dyck H. [2004]. A single indicator versus a multispecies approach: a case study on wet heathlands. In: Maes D., The use of indicator species in nature management and policy making. The case of invertebrates in Flanders [northern Belgium]. Institute of Nature Conservation, Brussels. pp. 184-206.

Het Instituut voor Natuurbehoud

Het Instituut voor Natuurbehoud [IN] is een wetenschappelijke instelling van de Vlaamse Gemeenschap; het telt momenteel een 100-tal medewerkers.

Het werd op 1 maart 1986 operationeel met als algemene taakstelling: “alle passende wetenschappelijke studies, onderzoeken en werkzaamheden uit te voeren in verband met het natuurbehoud, inzonderheid met het oog op het uitwerken van actiemiddelen en wetenschappelijke criteria tot het voeren van een beleid inzake natuurbehoud; hiertoe verzamelt het alle nuttige documentatie, onderneemt het de nodige studies en onderzoekingen, richt enquêtes in en zorgt voor de overdracht van de verworven kennis aan de bevoegde overheden...”

Het onderzoek heeft vooral betrekking op de diverse aspecten van de biodiversiteit, meer bepaald de inventarisatie, monitoring en ecologie van planten- en diersoorten, populaties en levensgemeenschappen in relatie tot hun omgeving. In het landschapsecologisch onderzoek gaat de aandacht vooral naar ecohydrologie, habitatfragmentatie en ecosysteemprocessen. De wetenschappelijke kennis ligt aan de basis van referentiekaders [zoals Rode Lijsten van diverse taxonomische groepen], karteringen van het natuurlijk milieu [zoals de Biologische waarderingskaart, BWK] en gebiedsgerichte acties inzake natuurontwikkeling, -herstel en -beheer. Dit beoogt het beleidsmatig inpassen van ruimtelijke en kwalitatieve noden van natuurbehoud in landinrichting, ruimtelijke planning, integraal waterbeheer en milieubeheer. Toepassingen liggen o.m. in de sfeer van het afbakenen van ecologische netwerken en gebieden van internationale betekenis en soortbeschermingsplannen.

Het Instituut is betrokken bij verschillende regionale, nationale en internationale onderzoeksprogramma’s en netwerken. Daarnaast is er nauwe samenwerking met universiteiten en andere wetenschappelijke instellingen in binnen- en buitenland.

Adviesverlening is een belangrijke taak van het Instituut. Deze gebeurt zowel ten behoeve van het Kabinet van de bevoegde Minister, de Vlaamse Hoge Raad voor Natuurbehoud , de Milieu- en Natuurraad van Vlaanderen, AMINAL, AHROM en andere entiteiten van de Vlaamse Gemeenschap.

In opdracht van derden kunnen via het Eigen Vermogen specifieke studies, karteringen en expertises worden uitgevoerd, waarvoor tijdelijke contractuele medewerkers kunnen worden aangetrokken.

Het Instituut voor Natuurbehoud publiceert rapporten en mededelingen in een eigen reeks. De bibliotheek biedt een ruim aanbod van tijdschriften en referentiewerken inzake milieu en natuur. Daarnaast biedt het Instituut diverse informatie aan via internet.

Algemeen Directeur: Prof. Dr. Eckhart Kuijken.

In Flanders, as in most other NW-European countries, decisions in nature conservation are often non-ecologically based. Species-specific information is, up-to-date, only rarely used in policy making or in evaluating or planning site selection or management. There is, however, a growing interest in using [indicator] species as tools or as goals in nature conservation in Flanders. Invertebrates constitute 75% of all biodiversity, but are often ignored as possible tools or goals in nature conservation. However, the fact that many invertebrates occupy narrow niches, use biotopes on a small scale, have a low mobility and react rapidly to changes in the environment, makes their 'information content' complementary to that of other better known species such as birds, mammals or plants. In this thesis, we demonstrate the surplus value of the use of [indicator] species [especially invertebrates] to nature conservation and policy making in Flanders.

Promotor: Prof. Dr. Eckhart Kuijken [Ghent University & Institute of Nature Conservation]
Co-promotor: Dr. Hans Van Dyck [University of Antwerp]
Date of public defence: 11 February 2004 [Ghent University]

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