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The use of opportunistic data for IUCN Red List assessments

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Abstract

IUCN Red Lists are recognized worldwide as powerful instruments for the conservation of species. Quantitative criteria to standardise approaches for estimating population trends, geographic range and population size have been developed at global and sub-global levels. Little attention has been given to the data needed to estimate species trends and range sizes for IUCN Red List assessments. Few regions collect monitoring data in a structured way and usually only for a limited number of taxonomic groups. Therefore, opportunistic data are increasingly used for estimating trends and geographic range sizes. Trend calculations use a range of proxies: i) monitoring sentinel populations, ii) estimating changes in available habitat or iii) statistical models of change based on opportunistic records. Geographic ranges have been determined using: i) marginal occurrences, ii) habitat distributions, iii) range-wide occurrences, iv) species distribution modelling (including site-occupancy models) and v) process-based modelling. Red List assessments differ strongly among regions (Europe, Britain and Flanders, north Belgium). Across different taxonomic groups, European Red Lists most often used IUCN criterion B and D. In Britain, criterion D and criterion A were the most frequently used, while in Flanders, this was the case for criterion B and criterion A. Among taxonomic groups, however, large differences in the use of the different IUCN criteria were revealed. We give examples from Europe, Britain and Flemish Red List assessments and give recommendations for a more uniform use of IUCN criteria among regions and among taxonomic groups.

Keywords: Britain, citizen science, Europe, Flanders (north Belgium), geographic range size, threatened species, trend calculations

Introduction

IUCN Red Lists are recognized worldwide as very powerful instruments for the conservation of threatened species (Lamoreux *et al.*, 2003; Rodrigues *et al.*, 2006). Although theoretically Red Lists are designed for estimating the extinction risk of species, they are used in conjunction with other information for setting priorities in the compilation of species action plans (e.g., Keller & Bollmann, 2004; Fitzpatrick *et al.*, 2007), reserve design and reserve management and as indicators for the state of the environment (Butchart *et al.*, 2006). The compilation of IUCN Red Lists has a long history (Scott, Burton & Fitter, 1987): the first assessments based on (subjective) expert opinion were produced in the 1970's for mammals (IUCN, 1972), followed by fish (IUCN, 1977), birds (IUCN, 1978), plants (Lucas & Synge, 1978), amphibians and reptiles (IUCN, 1979) and invertebrates (IUCN, 1983). Following recognition of the need to standardise approaches so as not to confound issues such as severity of threat and likelihood of extinction, more objective and quantitative criteria were developed in the 1990's (Mace & Lande, 1991; Mace *et al.*, 1993). These criteria have become widely implemented at the global (Mace *et al.*, 2008), national and regional level (Gärdenfors *et al.*, 2001; Miller *et al.*, 2007) as a means of classifying the relative risk of extinction of species.

As well as on the global level, Red Lists can also be compiled on continental (e.g., European, African), national (e.g., Eaton *et al.*, 2005; Keller *et al.*, 2005; Rodríguez, 2008; Brito *et al.*, 2010; Collen *et al.*, 2013; Juslén, Hyvärinen & Virtanen, 2013; Stojanovic *et al.*, 2013) or regional (sub-national) scales (e.g., Maes *et al.*, 2012; Verreycken *et al.*, 2014). Research has mainly focused on the implementation of the IUCN criteria at sub-global levels (Gärdenfors *et al.*, 2001), but far less attention has been given to the data needed and/or used to estimate species trends and rarity. As large and growing numbers of species are assessed at the global (76 000 species in the latest IUCN update) and sub-global level every year, greater scrutiny has necessarily been brought to bear on the types of data available to conduct such assessments (e.g., the latest update of the National Red List database contains 135 000 species assessments; www.nationalredlist.org).

Only few regions in the world collect data on trends, geographic range size and population sizes in a structured way (e.g., statistically sound monitoring networks – Thomas, 2005), usually for a limited number of taxonomic groups (e.g., birds – Baillie, 1990; butterflies – van Swaay *et al.*, 2008). Such data collection is often done with a network of volunteer experts (i.e., citizen science) under the co-ordination of professionals (e.g., Jiguet *et al.*, 2012). Monitoring data collected in a structured way allow for the use of most of the IUCN criteria, but require sustained funding (Hermoso, Kennard & Linke, 2014). Increasingly, opportunistic data (i.e., distribution records collected by volunteers in a non-structured way) are used for regional Red List assessments (e.g., Fox *et al.*, 2011; Maes *et al.*, 2012). Especially in NW Europe (Britain, the Netherlands, Belgium), the number of volunteers contributing to distribution and monitoring data is increasing yearly. In Flanders, for example, the online data portal www.waarnemingen.be of the volunteer nature ngo Natuurpunt started in 2008 and now has almost 20 000 active users submitting distribution records. The total number of records in the data portal at present amounts to more than 15 million, of which almost 2 million are accompanied by a picture to check identifications. Birds are by far the most recorded taxonomic group in Flanders (51%), followed by plants (26%), moths (8%), butterflies (5%), mushrooms (2%), mammals (2%), dragonflies (1%), beetles (1%), flies (1%), bees and wasps (1%), amphibians and reptiles (1%) and grasshoppers (1%). But, how suitable are these opportunistic data for Red Listing? Opportunistic data are often biased, both in time (e.g., recent periods are usually much better surveyed then 'historical' ones), in space (e.g., not all areas are surveyed with an equal intensity – Dennis, Sparks & Hardy, 1999), but also in volunteer preferences for taxonomic groups (e.g., birds, mammals, butterflies) and in differences in observation volunteer skills (e.g., identification errors, detectability - Dennis *et al.*, 2006). A growing diversity of approaches, however, has been developed to take these biases in opportunistic data into account when calculating trends in both abundance and in distribution and geographic ranges (Isaac *et al.*, 2014).

Here, we focus on opportunistic citizen science data used to classify species into IUCN Red List categories at sub-global levels (Gärdenfors *et al.*, 2001). We have reviewed the use of IUCN criteria in Europe, Britain and Flanders (north Belgium) and give examples of how they were applied in the different regions. Specifically, we examine the role of opportunistic data and compare them with data that have been collected in a standardized way for the assessment of population trends (IUCN criterion A) and for species' geographic range sizes (IUCN criterion B).

How red list assessments work: IUCN criteria and categories

Red List categories provide an approximate measure of species' extinction risk in a given region, by quantitatively evaluating some of the key symptoms of risk: 1) a trend in population size or distribution, 2) rarity (abundance) and/or restriction (geographic range) and 3) population size (number of reproductive individuals). These measures reflect the major determinants of risk identified by conservation biology (Caughley, 1994): species are at greatest risk of extinction when population size is small, decline rate is high and fluctuations are high relative to population growth. Very small populations are also more susceptible to negative genetic, demographic and environmental effects. At relatively large scale (e.g., global, continental), data are often very patchy (e.g., GBIF - Beck *et al.*, 2014), but this can also be the case on national or regional levels when survey intensity is low. The over-riding philosophy is to 'make do' with the available data, since the conservation problem is too pressing to wait for more robust data (Hermoso, Kennard & Linke, 2014). IUCN criteria are, therefore, designed to be used with different types of data (Mace, 1994).

The IUCN applies five main criteria to classify species in Red List categories:

A. Population size reduction
B. Geographic range size
C. Small population size and decline
D. Very small population or restricted distribution
E. Quantitative analysis of extinction risk.

Eleven IUCN categories are used for listing species in sub-global Red Lists (Fig. 1 – Gärdenfors *et al.*, 2001). These assessments use the same quantitative criteria as global Red Lists, but with an additional criterion to downgrade the risk category in cases where rescue effects, across national or regional borders are possible (Gärdenfors *et al.*, 2001). With opportunistic data, mainly IUCN criteria A (population trends) and B (geographic range sizes) can be estimated.

IUCN criterion use in Europe, Britain and Flanders

First, we review the use of the different IUCN criteria for Red List assessments in three 'regions': Europe (continental), Britain (national) and Flanders (north Belgium - regional). We also give examples of appropriate methods to estimate trends and geographic range sizes for regional Red List assessments. The list of IUCN Red Lists screened is given in Table 1.

The proportions of the different criteria used over all taxonomic groups in Europe, Britain and Flanders are given in Fig. 2. For the European Red Lists, the most frequently used criteria were B (57%) and D (32%). In Britain, criterion D (47%) and criterion A (27%) were the most frequently used, while in Flanders, this was the case for criterion B (57%) and criterion A (25%). Among taxonomic groups, however, large differences in the use of the different IUCN criteria were revealed (Fig. 3). In Europe, criterion A was mainly used for classifying mammals (44%) and butterflies (43%), criterion B for saproxylic beetles (85%), amphibians

(68%) and reptiles (63%), criterion C for dragonflies (21%) and criterion D for terrestrial (51%) and freshwater molluscs (39% − Fig. 3). In Britain, criterion A was mainly used for classifying butterflies (67%) and plants (44%), criterion B for dragonflies (100%) and water beetles (80%), criterion C for flies (30%) and criterion D for boletes (100%) and lichens (68% − Fig. 3). In Flanders, criterion A was mainly used for classifying waterbugs (50%), freshwater fishes (29%) and ladybirds (27%), criterion B for reptiles (100%) and amphibians (83%), criterion C for mammals (18%) and amphibians (17%) and criterion D for mammals only (44% − Fig. 3).

Proxies for population trend estimates

Few species globally have their entire population monitored regularly in order to properly assess trends in population size. One of several shortcuts is, therefore, typically employed. A first possible shortcut is to use a small number of sentinel populations that are monitored regularly, either at long-term research sites or as part of co-ordinated schemes such as the UK or Dutch Butterfly Monitoring Scheme (Botham *et al.*, 2013; van Swaay *et al.*, 2013) or the Breeding Bird Survey in the UK or Flanders (Harris *et al.*, 2014; Vermeersch & Onkelinx, 2014). This approach can deliver precise trend estimates, but in most cases the populations are a biased subset and may not be representative of the wider species' population (Brereton *et al.*, 2011). A second and coarser tool is to estimate changes in the amount of available habitat, typically from polygon maps, but problems with this approach (commission and omission errors, see further) have been documented and discussed (Boitani *et al.*, 2011). The approach is appealing, as remote sensed data on change in habitat extent can be cost-effectively applied to a range of species. However, even if changes in habitat can be captured accurately, it is unclear how trends reflect actual trends in abundance (Van Dyck *et al.*, 2009). Thus, both these proxies rely on a large number of untested assumptions. A third proxy is to construct a statistical model of change based on opportunistic biological records. Often, measures of change from biological records have been derived from simple 'grid cell counts' between atlas periods (e.g., Maes & van Swaay, 1997; Maes & Van Dyck, 2001; Thomas *et al.*, 2004; Maes *et al.*, 2012), which is conceptually similar to the use of habitat extent maps described above. Estimating change from biological records is complicated, because the intensity of recording varies in space and time (Prendergast *et al.*, 1993; Isaac & Pocock, this volume) and can be difficult to estimate from the records alone (Hill, 2012). The development of methods for estimating trends from biological records has recently been the subject of considerable research effort and several robust approaches are increasingly being used. Abundance data is generally considered superior to distributional data for trend estimation and statistical methods are starting to be developed which derive composite trends using models that combine information from both data types (Pagel *et al.*, 2014).

Estimating population trends with opportunistic data

Using the IUCN criteria, a population trend (criterion A) can be assessed in five different ways: (Aa) direct observation, (Ab) an index of abundance, (Ac) a decline in the area of occupancy (AOO), the extent of occurrence (EOO) or habitat quality, (Ad) actual or potential levels of exploitation or (Ae) effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites. The use of criterion Ab (an index of abundance) depends strongly on the taxonomic group (e.g., for British butterflies, an index of abundance (criterion Ab) is available for 49 out of 62 resident species (79%), Fox *et al.*, 2011 – Box 1). Trends are most often calculated using changes in the AOO or the EOO or in habitat quality (criterion Ac – Box 2), especially in Britain (93%) and in Flanders (91% − Fig. 4). In Europe, trends calculated as a decline in distribution area and/or habitat quality (criterion Ac) are used in 50% of the cases. The effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites (Ae) is used in 22% of the cases (Fig. 4). Criterion Ae was used mainly for freshwater organisms such as fishes and molluscs where invasive species are a major problem (Strayer, 2010; Roy et al, this issue). In Flanders, this criterion was also used for the negative effect of the Harlequin ladybird on native ladybirds (Roy *et al.*, 2012a).

Box 1 – Trend calculations using abundance data from standardized citizen science monitoring data (IUCN criterion Ab)

There is a wide spectrum of citizen science approaches which contribute to monitoring biodiversity, ranging from simple protocols with wide participation to structured approaches which often include elements of profession support and co-ordination (Schmeller *et al.*, 2009; Roy *et al.*, 2012b; Pocock & Isaac, this volume). Structured, participatory monitoring schemes such as those established for birds, butterflies and mammals in Europe and North America (Devictor, Whittaker & Beltrame, 2010) typically comprise counts of target species throughout the year, repeated annually at fixed locations across a region. For example, the UK Butterfly Monitoring Scheme (UKBMS) provides a standardised annual measure (index) of butterfly populations at line-transect sites (Rothery & Roy, 2001). The UKBMS was initiated in 1976 with 34 sites, rising to more than 100 sites per year from 1979 onwards and currently comprises 2000 sites recorded annualy. The UKBMS also incorporates a Wider Countryside Butterfly Scheme component to improve the spatial coverage of the scheme (Roy *et al.*, 2015) Indices from different UKBMS sites over years are combined to derive regional and national collated indices, which can be used to assess long- and short-term population trends (Pannekoek & van Strien, 2001). The UKBMS has been used to assess threat status of 49 out of 62 species (79%) over two time periods: (i) 10 years (1995–2004) and (ii) long-term (typically 1976–2004) for the Red List of British Butterflies (Fox *et al.*, 2011). Other examples of the use of structured monitoring schemes are the bird scheme in the UK where 22 out of 74 species (30%) were classified as threatened on the basis of trends in abundances (Eaton *et al.*, 2005).

One advantage of volunteer-based, structured monitoring scheme is good statistical power for measuring trends (e.g. Roy, Rothery & Brereton, 2007) and capacity to generate time series with comprehensive spatial coverage of a region. They have also provided a rich resource for scientific research, investigating large-scale pattern and processes (Thomas, 2005). Although there has been a growth in the number of such schemes in some regions (N America and NW Europe) during the current century (Nature Editorials, 2009), there remains a paucity for many species groups in most parts of the world. Successful schemes often rely on institutional support and funding, as well as having a large pool of potential contributors. Although we recommend adopting best practice from established schemes to further their value for future Red List criterion Ab assessments, distribution data is typically available for a wider set of species groups and for more regions of the world (see Box 2).

Box 2 – Trend calculations using opportunistic distribution data (IUCN criterion Ac)

Citizen science data are a potentially valuable source of information on distribution changes, but they suffer from uneven and unstandardized observation effort (Isaac & Pocock, this volume). Changes in observation efforts across years may easily lead to artificial trends or mask existing trends in species' distributions.

In the past, researchers used broad time periods in their comparisons of distribution to ensure sufficient effort and spatial coverage in each time period (van Swaay, 1990). Other authors have filtered their data and used thresholds of completeness of sampling per grid cell (cf. Soberón *et al.*, 2007) for estimating trends (e.g., Maes *et al.*, 2012). Recently, the methods available for trend estimations have developed substantially (Powney & Isaac, this volume). Isaac *et al.* (2014) tested a number of approaches for estimating trends from noisy data. Using simulations they found that simple methods may easily produce biased trend estimates, and/or had low power to detect genuine trends in distribution. Two sophisticated methods known as Frescalo and site-occupancy models emerged as especially promising. Frescalo uses information about sites' similarity to neighbouring sites to assign local benchmark species (Hill, 2012). These benchmarks provide a measure of local observation effort that can be statistically corrected. Frescalo was used to assess changes in plant species distributions for the recent Red List of vascular plants in England (Stroh *et al.*, 2014).

Site-occupancy models have a special mechanism to adjust for observation effort. They separate occupancy (the presence of a species in a site) from detection (the observation of the species in that site) when analysing field survey data (MacKenzie *et al.*, 2006). The models require that species are recorded as an assemblage, such that observations of one species can be used to infer non-detection of others (Isaac & Pocock, this volume). Detection can be estimated from sites that were surveyed multiple times in any given time period (e.g., a year). If observation effort increases over time, a species will be observed during more visits, which leads to a higher detection probability, but not to a higher occupancy probability (van Strien, van Swaay & Termaat, 2013). Site-occupancy models have been used in status assessments of butterflies and dragonflies in the Netherlands (van Strien *et al.*, 2010; van Strien, van Swaay & Termaat, 2013).

Methods for estimating geographic range size

Geographic range can be expressed in two ways according to the IUCN criteria: extent of occurrence (EOO – criterion B1) and area of occupancy (AOO – criterion B2). The EOO is defined as the area contained within the shortest continuous imaginary boundary which can encompass all the known, inferred or projected sites of present occurrence of a taxon, excluding cases of vagrancy. The AOO is defined as the area within its extent of occurrence, which is occupied by a taxon, excluding cases of vagrancy. IUCN guidelines advocate the use of 2 x 2 km² grid cells to estimate area of occupancy (IUCN Standards and Petitions Subcommittee, 2013). Geographic ranges can be determined using different approaches: i) marginal occurrences, ii) habitat distributions, iii) range-wide occurrences, iv) species distribution modelling (including site-occupancy models) and v) process-based modelling (Gaston & Fuller, 2009). Marginal occurrences, i.e., mapping the outer boundaries of species and subsequently interpolating the area in between, typically result in overestimated distribution ranges. Such maps are often displayed in field guides to illustrate the possible species distribution range in a usually large region (e.g., world, continent – Graham & Hijmans, 2006). A second way to estimate the geographic range of species is to use its habitat and/or associations with environmental variables as a proxy (Boitani *et al.*, 2011). When range-wide occurrences are available for a focal region (country), records are often assigned to a grid cell projection (e.g., Universal Transverse Mercator – UTM) to produce local or regional distribution atlases. At fine resolution (e.g., 1 x 1 km² or 5 x 5 km²), these data are sufficient to capture a species' distribution, so long as sampling intensity is spread over the region (Gaston & Fuller, 2009). Coarse grid cells (e.g., 10 x 10 km² or even 50 x 50 km²) are seldom useful for regional conservation purposes, because they include too much unsuitable habitat (Rondinini *et al.*, 2006), but recently, downscaling methods have been proposed to estimate local occupancy from coarse-grain distribution atlas data (Barwell *et al.*, 2014). Species distribution modelling is a helpful tool to determine a species geographic range (Pena *et al.*, 2014). Typically, presence/absence or presence-only data are used in different modelling techniques (Guisan *et al.*, 2013) to 'predict' where suitable environmental conditions occur in a given region for a given species (e.g., Thomaes, Kervyn & Maes, 2008; Cassini, 2011; Syfert *et al.*, 2014). Since site-occupancy models produce predicted occurrence probabilities per grid cell, they also permit to estimate geographic range sizes (either AOO or EOO, depending on the resolution of the grid cells) by summing the area of the grid cells for which species presences were predicted (van Strien, van Swaay & Kéry, 2011). Finally, processed-based modelling using small-scale environmental variables (e.g., microclimate) can be used to determine the possible geographic range of species (e.g., Kearney, 2006; Kearney *et al.*, 2014; Tomlinson *et al.*, 2014; Panzacchi *et al.*, 2015). Range-wide occurrences tend to underestimate the geographic range of species due to incomplete sampling (omission errors), while the other approaches tend to overestimate the distribution range of species (commission errors) because it incorporates large areas in which the species cannot occur (Gaston & Fuller, 2009).

Estimating geographic range sizes with opportunistic data

The geographic range of a species can be calculated in two different ways: the extent of occurrence (B1 – EOO) and/or the area of occupancy (B2 – AOO). Europe and Britain used criterion B1 and B2 together, while Flanders only used either criterion B1 or B2 (Fig. 4). In Europe, the joint use of both criterion B1 and B2 (50%) and criterion B2 separately (50%) were used equally often, probably depending on individual species' data availability. In Britain, the simultaneous use of criterion B1 and B2 was largely preferred (76%), while in Flanders the area of occupancy (criterion B2) was used in the majority of cases (86% - Fig. 4). In regions where the spatial coverage of opportunistic data is low (cf. Adriaens *et al.*, 2014, submitted), the extent of occurrence (criterion B1) is mostly used to estimate geographic range size (Box 3), while the area of occupancy (criterion B2) is more often used when the focal region has a high spatially mapping coverage.

Box 3 Estimating Extent of Occurrence (EOO, criterion B1)

Ecological ecodistricts for ladybirds in Flanders (north Belgium)

The extent of occurrence (EOO) is defined as the area contained within the shortest continuous imaginary boundary which can encompass all the known, inferred or projected sites of present occurrence of a taxon, excluding cases of vagrancy. For some regions and for particular taxonomic groups, opportunistic data are available on a high resolution and covering a large part or even the entire region (e.g., birds in the UK – Balmer *et al.*, 2013; butterflies in Flanders – Maes *et al.,* 2012). In such cases, the regional geographic range size is the sum of the area of these high resolution grid cells in which a species was observed in a recent period (e.g., 1 x 1 km² − Maes *et al.*, 2012 or 2 x 2 km² − Fox *et al.*, 2011). In regions where mapping coverage is fairly small, however, the area of occupancy (AOO) will be strongly underestimated using the sum of the area of high resolution grid cells. Here, the use of the EOO instead of the AOO is advocated to estimate the geographic range size. For ladybirds in Flanders, for example, the use of EOO is favoured as a measure of geographical range rather than the AOO. The rationale behind this is that the *range fill* for ladybirds is much smaller than for well-covered taxonomic groups such as butterflies. For ladybirds, on average, 5.7 grid cells of 1 x 1 km² were surveyed per 5 x 5 km² grid cell in the period 2006-2013 resulting in a relatively small *range fill*. For comparison, the average number of surveyed 1 x 1 km² grid cells per 5 x 5 km² grid cell for butterflies was 17.2, a *range fill* that is three times as high as for ladybirds (Maes *et al.,* 2012). The sum of all occupied 1 x 1 km² grid cells as the AOO would, therefore, largely underestimate their geographic range size. As geographic range size for ladybirds in Flanders, we, therefore, used the sum of the areas of the ecological districts (n = 36, Fig. 5) when the species was observed in at least three 1 x 1 km² grid cells in the period 2006-2013. These ecological districts are homogeneous with respect to abiotic characteristics that are relatively constant across time (e.g., climatology, geology, relief, geomorphology) and have similar landscape, soil and biotope types (Couvreur *et al*., 2004). The minimum number of three grid cells per ecological district was applied to exclude single observations of vagrant or erratic individuals. (Adriaens *et al.*, 2014, submitted).

Minimum Convex Polygons for plants and bees in the UK

One of the simplest method to estimate a species' extent of occurrence (EOO) is to calculate the Minimum Convex Polygon (MCP), the smallest polygon that will contain all the points and in which no internal angle is greater than 180 degrees (Fig. 6b). The MCP has, however, been criticised as being sensitive to errors in location, being derived from the most extreme points (Burgmann & Fox, 2003) and for incorporating large areas of unsuitable habitat. Two alternative methods to calculate species ranges that are less susceptible to these issues are: 1) the α-hull (Burgmann & Fox, 2003) and 2) the Localised Convex Hulls (LoCoH) (Getz & Wilmers, 2004). Both of these methods have recently been applied to Red List assessments in the UK for vascular plants (Stroh *et al.*, 2014) and aculeate Hymenoptera (www.bwars.com; Edwards *et al.*, in prep).

The α-hull is derived from a mathematical algorithm for converting points (the locations of records) into triangles based on a threshold parameter α (Burgmann & Fox, 2003). The hull produced becomes more inclusive and approaches the MCP as α increases (Fig. 6c).

The Localised Convex Hull (LoCoH) is an adaptation of the MCP but rather than fitting one hull to the entire dataset, the LoCoH is the result of the union of a set of 'localised' MCPs created by fitting the MCP to subsets of the data (Getz & Wilmers, 2004). There are several ways in which these local subsets can be determined (Getz *et al.*, 2007): 1) fixed number of points (*k*-LoCoH) in which subsets consist of *k*-1 closest points to each root point, 2) fixed sphere-of-influence (*r*-LoCoH) in which subsets consist of all points within a radius *r* of each root point, and 3) adaptive sphere-of-influence (*a*-LoCoH) in which subsets consist of the root point and the closest points where the sum of the distances between the points in the subset and root is less than *a*. In the UK Red Listing exercises for plants and aculeate Hymenoptera, the fixed sphere-of-influence method (r-LoCoH) was used as it facilitated the data review for the taxonomic exports and becasue it gave a visual understanding of the final Red Listing decisions (Fig. 6d). This variant of LoCoH should also be relatively robust to sporadic but spatially clustered recording and the presence of duplicate records both of which are relatively common in opportunistic citizen science data.

In both the α-hull and LoCoH, the resulting area is dependent on the value of a control parameter (α for α-hull and *k*, *r*, or *a* for the LoCoH variants). The selection of this parameter is a non-trivial process as it has a marked impact on the EOO estimates. Conceptually, there is no 'correct' value. Rather, the most suitable value depends upon a) the aims of the study, i.e., a trade-off between being as inclusive as possible at the cost of including some unsuitable areas (commission errors) or being cautious at the cost of excluding of some suitable areas (omission errors), b) the degree of spatial coverage in the data (with poorly sampled data requiring higher parameter values) and c) the properties of the taxa being investigated (e.g., for highly mobile taxa, the most appropriate value is larger than for sedentary ones while large values for linearly distributed taxa (e.g., coastal species) can result in the incorporation of large areas of unsuitable habitat). In the UK Red Listing exercises mentioned above the parameter values were selected to match the IUCN guidelines and previous Red Listing exercises (i.e., vascular plant) on the one hand or through expert opinion based on the outputs produced using a series of parameter values on the other.

Discussion

IUCN enables the use of five different criteria to estimate the extinction risk of species: A) population size reduction, B) geographic range size, C) small population size and decline, D) very small population and/or restricted distribution and/or E) quantitative analysis of extinction risk. In the ideal case, the presence of a statistically sound monitoring scheme in a focal region would allow the use of all IUCN criteria to assess the Red List status of species. Gathering standardised data for a large number of taxonomic groups and in a sufficiently large number of sites to be representative for the region, however, require sustained funding (Hermoso, Kennard & Linke, 2014). With opportunistic data, only IUCN criterion A and B can be used, because criteria C, D and E clearly need more standardized data to estimate population sizes and structures. For criterion D, however, opportunistic data can be used to estimate a very small area of occupancy (<20 km²) and/or a very small number of populations, given that the mapping intensity is sufficiently high. In addition, expert opinion of citizen scientists and or professionals can be used to estimate population size classes (e.g., <250, 250-2500, 2500-10 000 for criterion C) of some relatively well-known taxonomic groups (e.g., mammals, birds).

How many (opportunistic) data are needed to calculate population trends (criterion A) and geographic range sizes (criterion B) for an IUCN Red List assessment? In Flanders, prior to the compilation of an IUCN Red List, the institute co-ordinating all regional Red List assessments (i.e., the Research Institute for Nature and Forest - INBO) applies a quantitative and simple procedure to judge whether the dataset contains enough data to reliably estimate trends and range sizes. First, the Red List compilers are asked to decide which periods will be compared to calculate population trends. Here, IUCN recommends a recent period of 10 year or three generations, whichever is the longer (IUCN, 2003), but many Red List compilers use historical periods that are longer than 10 years usually to compensate for the lower number of historical records in many data sets (e.g., the English Red List of plants – Stroh *et al.*, 2014). Second, for these periods, the grid cells that have been sufficiently well mapped in common in both periods are located. Mapping intensity can be estimated using species completeness measures (Soberón *et al.*, 2007), rarefaction measures (Carvalheiro *et al.*, 2013), reference species (Maes & van Swaay, 1997) etc. In a third step, the sufficiently well-surveyed grid cells are attributed to the twelve ecological regions in Flanders that have similar biotopes, soil types and landscapes (Couvreur *et al.*, 2004). To make a representative Red List for a focal region, the recommendation for Flanders is that distribution data should be available in a minimum number of the grid cells (e.g., 5%) in all the (relevant) ecological regions for the given taxonomic group. If a data set of a taxonomic group does not fulfil these criteria, it is considered as currently insufficient for the compilation of an IUCN Red List in Flanders. Fig. 7 visualizes this procedure for dolichopodid flies and butterflies, the first group failing to pass, while the latter does.

On larger scales (e.g., world, continental, European Union), it would be biologically more meaningful to make Red Lists per ecological and/or biogeographical regions as, for example, for the global biodiversity hotspot of the Mediterranean region (Myers *et al.*, 2000). In this region, such lists have been compiled for mammals (Temple & Cuttelod, 2009), dragonflies (Riservato *et al.*, 2009), freshwater fishes (Smith & Darwall, 2006), cartilaginous fishes (Cavanagh & Gibson, 2007) and amphibians and reptiles (Cox, Chanson & Stuart, 2006). On the other hand, conservation planning is usually the responsibility of national governments, which makes biogeographical Red Lists difficult to apply in the field.

Due to differences in scale requirements and longevity among species (e.g., short-lived invertebrates versus long-lived vertebrates), but also because of differences in data availability, some have argued that IUCN criteria should be differentiated for taxonomic groups (e.g., invertebrates – Cardoso *et al.*, 2011; Cardoso *et al.*, 2012) and/or for spatial scales (Brito et al., 2010). Some countries continue to use national Red List criteria and categories instead of those of the IUCN criteria because they judge them unusable in smaller regions (e.g., the Netherlands – de Iongh & Bal, 2007). If applied correctly and even with the use of opportunistic and/or data, we are convinced that the present-day IUCN criteria can be used for a wide variety of taxonomic groups, including invertebrates (Collen & Böhm, 2012) and at many different spatial scales (from global to regional). The key point is that such data should be scrutinised and not used blindly. IUCN Red Lists are useful to countries or regions since they need to understand and track the fate of species within their borders. Legislation such as the Convention on Biological Diversity encourages countries to do this at a national level (Zamin *et al.*, 2010). The interesting ones are where there are great discrepancies. For example, should Britain care about a butterfly species that is at the edge of its northern range in a restricted area within the south of the region? From a global or continental extinction risk perspective, probably not. The vast population in the rest of mainland Europe means that the potential loss of the species in Britain is no threat to its overall survival. Since the butterfly is part of Britains biodiversity and is considered nationally threatened, however, it should be protected and conserved. This clearly demonstrates the difference between a Red List which ‘only’ estimates the extinction risk of a given species in a focal region on the one hand and a national or regional list of conservation priorities on the other (Lamoreux *et al.*, 2003). Red Lists should, therefore, be considered is a decision *support* tools and not as decision *making* tools (Possingham *et al.*, 2002).

To conclude, we give some recommendations that may help to apply IUCN criteria more uniformly across taxa and across regions from an organisational point of view but also for peers that compile Red List in other parts of the world. Documenting a Red List assessment is of vital importance to understand trend analyses and geographic range size estimates. Therefore, it is important to document spatial and temporal mapping intensity in the focal region, to give detailed information on how trends, distribution ranges and population sizes were calculated and which assumptions were made in the analyses. Important organisational aspects that can improve Red List assessments are, among others, the assignment of a Red List co-ordinator in a region to have consistency among Red Lists of different taxonomic groups (e.g., BRC in Britain, the Research Institute for Nature and Forest (INBO) in Flanders), the availability of the dataset used for the Red List assessment for peers (open access data, e.g., GBIF, National Red List database; www.nationalredlist.org), the motivation and documentation of expert-judgement when using subcriteria such as fragmentation, fluctuations and rescue effects or for the estimation of population sizes.

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Table 1.
IUCN Red Lists in Europe, Britain and Flanders that were screened on the use of the different IUCN criteria.

Europe (ec.europa.eu/environment/nature/conservation/species/redlist/)
Amphibians (Temple & Cox, 2009); Butterflies (van Swaay *et al.*, 2010); Dragonflies (Kalkman *et al.*, 2010); Freshwater fishes (Freyhof & Brooks, 2011); Freshwater molluscs (Cuttelod, Seddon & Neubert, 2011); Mammals (Temple & Terry, 2007); Reptiles (Cox & Temple, 2009); Saproxylic beetles (Nieto & Alexander, 2010); Terrestrial molluscs (Cuttelod, Seddon & Neubert, 2011); Vascular plants, partim (Bilz *et al.*, 2011)
Britain (jncc.defra.gov.uk/page-3352)
Boletes (Ainsworth *et al.*, 2013); Butterflies (Fox, Warren & Brereton, 2010); Dragonflies (Daguet, French & Taylor, 2008); Flies (Falk & Crossley, 2005; Falk & Chandler, 2005); Lichens and lichenicolous fungi (Woods & Coppins, 2012); Vascular plants (Cheffings *et al.*, 2005); Water beetles (Foster, 2010)
Flanders (www.inbo.be/content/page.asp?pid=BEL_VLA_SOO_rodelijstIUCN)
Amphibians (Jooris *et al.*, 2012); Butterflies (Maes *et al.*, 2012); Freshwater fishes (Verreycken *et al.*, 2014); Ladybirds (Adriaens *et al.*, 2014); Mammals (Maes *et al.*, 2014); Reptiles (Jooris *et al.*, 2012); Stag beetle (Thomaes & Maes, 2014); Water bugs (Lock *et al.*, 2013)

Figures

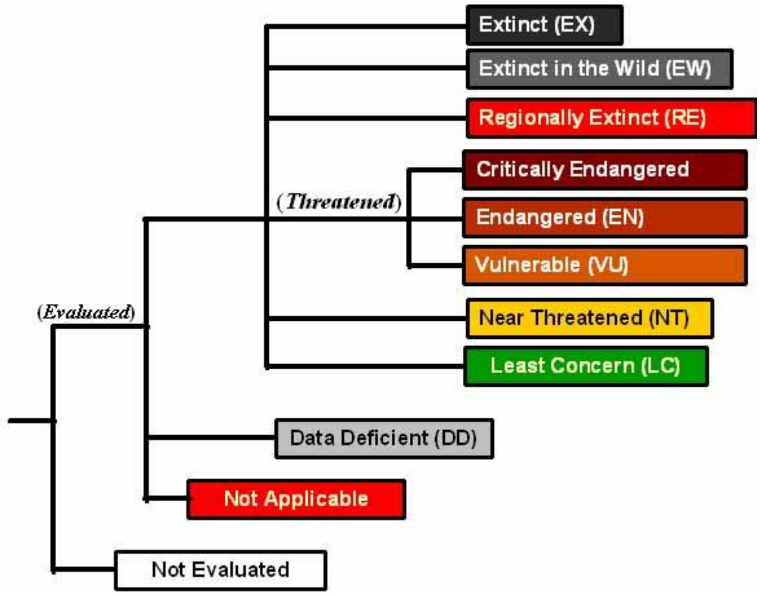

Figure 1.
IUCN categories at the regional level (IUCN, 2003).

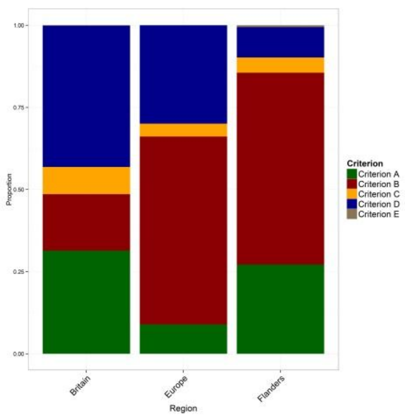

Figure 2.
Overall criterion use for species in Britain (total number of threatened species = 1569), Europe (n = 714) and Flanders (n = 125). Criterion A = Population size reduction, Criterion B = Geographic range size, Criterion C = Small population size and decline, Criterion D = Very small or restricted population, Criterion E = Quantitative analysis of extinction risk.

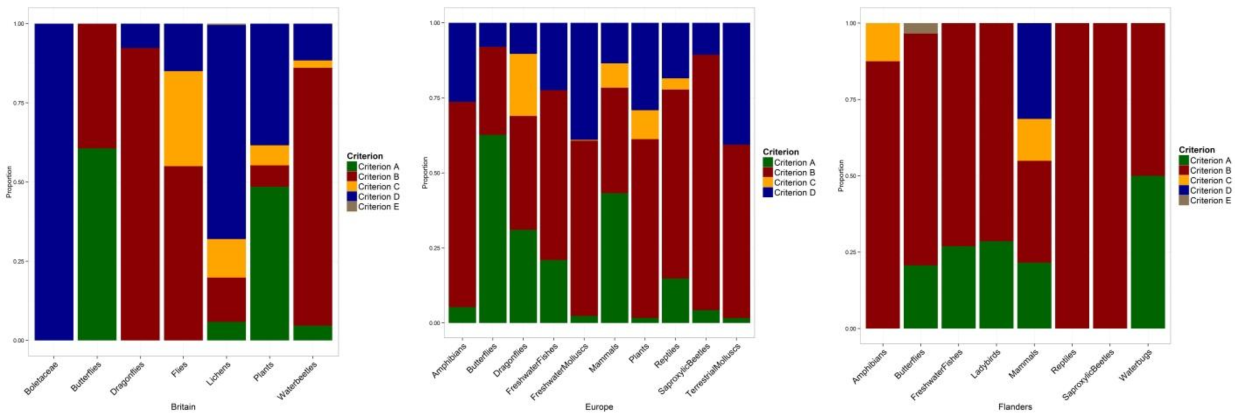

Figure 3.
Criterion use per taxonomic group in Britain (left), Europe (middle) and Flanders (right).

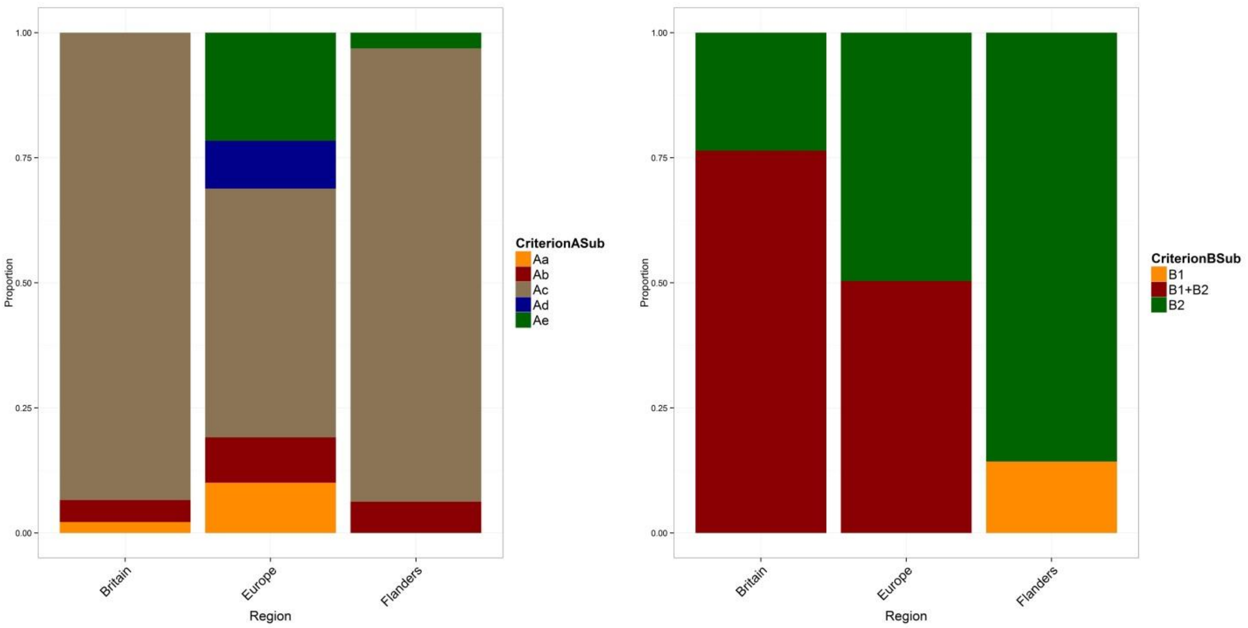

Figure 4.
Use of approaches in IUCN criterion A (population size reduction, left) and IUCN criterion B (geographic range size, right) in Red List assessments in Britain, Europe and Flanders. Criterion A: Aa = direct observation, Ab = an index of abundance appropriate to the taxon, Ac = a decline in area of occupancy, extent of occurrence and/or habitat quality, Ad = actual or potential level of exploitation, Ae = effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites; Criterion B: B1 = extent of occurrence, B2 = area of occupancy, B1+B2 = extent of occurrence + area of occupancy.

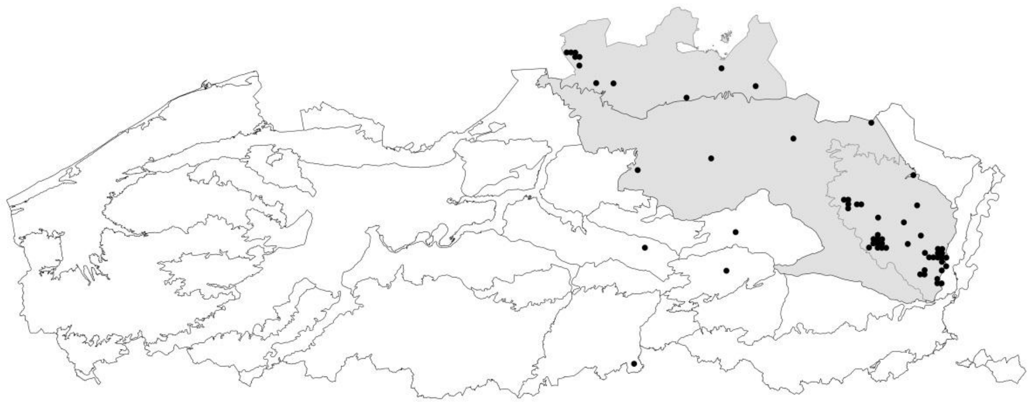

Figure 5**.**
Extent of occurrence (EOO) of the ladybird *Coccinella hieroglyphica* using the 36 ecological districts in Flanders (north Belgium) in the period 2006-2013. The distribution of the species is shown using 1 x 1 km² grid cells (black dots). Only ecological districts (in grey) in which the species was observed in at least three grid cells were incorporated in the estimate of the extent of occurrence (i.e., 3 087 km² – Adriaens *et al.*, 2014, submitted).

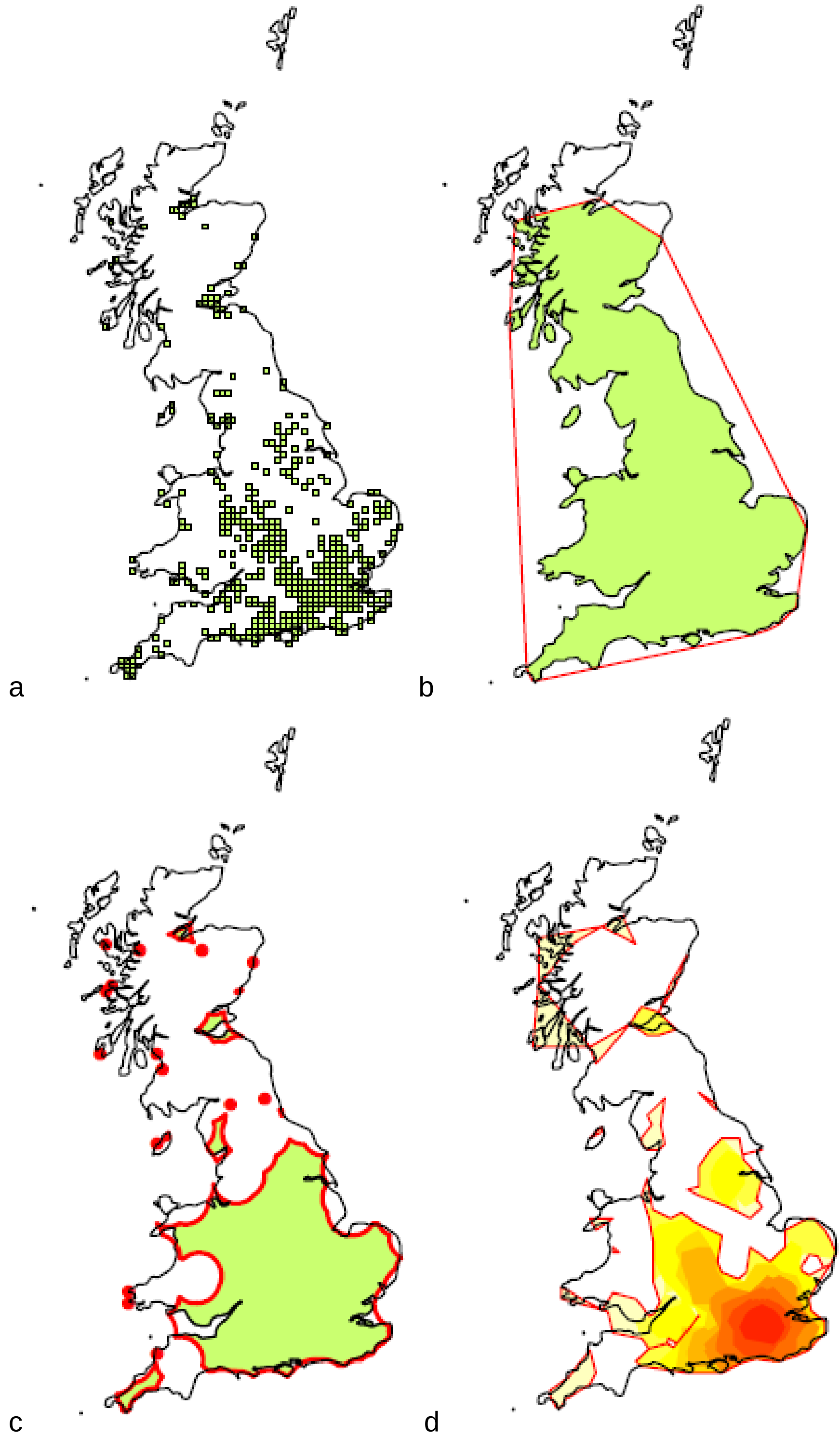

Figure 6.

Maps showing the extent of occurrence (EOO) estimates for the UK from the period 1996-2010 for the bee *Andrena bicolor* using a) observed 10 x 10 km² grid squares (total area = 46 100 km^2), b) Minimum Convex Polygon (MCP – 324 850 km^2 for full MCP or 208 150 km^2 for intersection of MCP with land area) c) α-hull (101 895km^2) and d) r-LoCoH (101 919 km^2). These figures were produced for a Red Listing assessment of aculeate Hymenoptera in Great Britain (Edwards *et al.*, in prep).

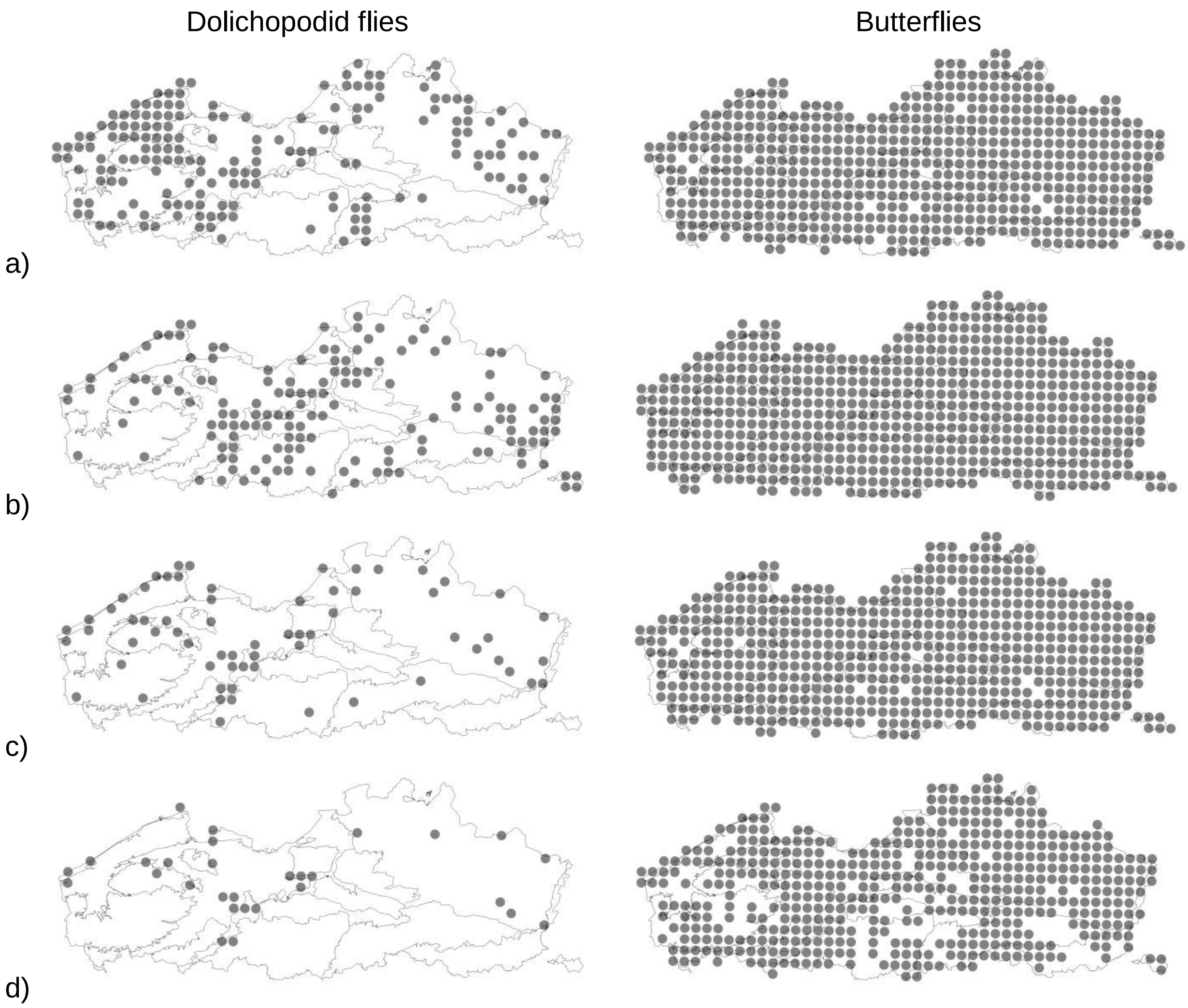

Figure 7.
Visualization of the procedure used in Flanders (north Belgium) to judge whether enough data are available for a Red List assessment. As a background, the 12 ecological regions of Flanders are shown. a) all grid cells (5 x 5 km²) surveyed in the first period for dolichopodid flies (left) and butterflies (right), b) all grid cells surveyed in the second period, c) all grid cells surveyed in common in both periods, d) all grid cells in common in both periods that are considered as sufficiently well surveyed (i.e., ≥ 10 species per grid cell in both periods).