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Report on a possible implementation of a Community Specialisation Index at European level: an indicator of biodiversity state and ecosystem integrity

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Executive Summary

Some elements of concept

1. Ecosystems under stress such as habitat loss, fragmentation or other disturbance including pollution suffer from a decline of specialist species while generalist species could increase (Marvier, 2004), which resulted in “biotic homogenization process” (communities, ecosystems tend to become more and more similar across spatial scales)

2. A Community Specialisation Index (CSI) can be a good proxy to quantify this biotic homogenization process and to reflect the state and health of biodiversity and ecosystems.

3. There is strong conceptual evidence that Community Specialisation, that reflects the relative composition between specialist species with a narrow ecological niche and generalist species which are less requiring, reflects ecosystem stability and so in large part ecosystem integrity.

Evaluation of Species Specialisation

4. Species specialisation expressing the preferences of a species for one or several habitats can be evaluated from different data sources (Expert European database versus standardized collected data from monitoring). There is a significant correlation between the two calculations. From a practical aspect, evaluation of species specialisation based on expertise is less precise but easier than based on monitoring information which is often not available.

5. However, a specialisation of one species could differ with the localisation within its distribution range area, better assessing relationship between species and habitat at regional scale should be helpful to gain sensitivity of the indicator.

Sensitivity of Community Specialisation Index to spatial change in Europe

6. Based on a spatial perspective, analysis of sensitivity of the CSI at European scale shows that it is, first of all, driven by geoclimatic variables. In a second step, it is possible to show its sensitivity facing pressures even using data of “poor sensitivity” such as presence-absence of species in cell of 50 km x 50 km. Results show there are some very strong relationships between pressures upon biodiversity and CSI for taxa groups with distribution not massively driven by biogeographical and/or thermal factors.

Feasibility of using CSI trends across time in Europe

7. Analyses of GBIF results in comparison with known trends tend to show that GBIF data can be used to study spatial structures.

8. But for a temporal analysis, the use of GBIF database does not allow highlighting CSI trend for birds in Europe during the five last decades such as it does not allow at this time finding known temporal trends obtained using standardized collected data.

9. Since the lack of information especially concerns temporal trends, these results tend to limit the usability of current GBIF data to highlight temporal trends of biodiversity under global changes through CSI approach.

10. Note that even if the database seems to be large, in reality, it does not contain at this time the main part of available databases on Biodiversity.

11. Using data from national standardized monitoring, a significant decline of CSI at European scale through the two last decades can be shown as an evidence of the ongoing functional homogenization process of biodiversity.

Trend of CSI driven by changes in anthropogenic pressures

12. Analysing how the decline is explained by land–use pressures changes, it appears to be worth using information on pressures at small scale (1km²).

13. As expected, the bird communities are significantly more specialized where farmland of high nature value are dominant with a higher CSI in proportion of HNV area.

14. But surprisingly, the CSI declined more in zones dominated by HNV farmland. This effect could be due to the initial higher values of CSI in HNV zones and by definition, the CSI of specialized communities can decrease stronger than the CSI of generalized communities Therefore the trend can be stronger where CSI is higher. This trend could also be explained by an increase of generalists due to the withdrawal of agriculture within these zones.

15. Further studies are needed to improve the analyses and better understand the observed trends. But, many evidences showed the relevancy of using the CSI approach to survey biodiversity state and ecosystem integrity under global changes at large scale.

Recommendations

16. CSI proves to be a very good indicator for biodiversity status. It is reactive to changes and much more informative than classic indicators based on species richness or species abundance variations as it encompasses both biodiversity structure and functionalities.

17. In order to progress further on this indicator, it is suggested to:
    - explore the variability of species specialisation across their biogeographic range and then develop regional databases of species preferences per habitat to gain in robustness of the indicator.
    - develop the approach for other taxonomic groups (butterflies, plants), and examine the relevance of the approach for marine and freshwater taxonomic groups.
    - continue to test the sensitivity of the indicator to drivers such as land-use changes, using CORINE Land-Cover 1990, 2000 and 2006, using for example birds species abundance data in different European countries, at different spatial

scales and in complement of others indicators (richness, MSA and other MSTA such as those based on trophic levels).

18. If the development of such sensitive and robust indicators is crucial, it is also urgent to develop, promote, support standardized monitoring programs of biodiversity on a large spatial scale that allow comparisons across time and space and can therefore effectively monitor the status of biodiversity.

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Aim of the report:

The present report was prepared as part of the European Topic Centre on Biological Diversity's mandate to contribute to framing the development of a European Biodiversity Index. It follows-up the production, in June 2009, of a report to the European Environment Agency, entitled "Working paper on a Community specialisation index" (Loïs et al, 2009), which have begun to explore the feasibility of Community Specialisation Index (CSI), as a proxy to reflect a certain level of ecosystem integrity.

Here, the document presents after a brief report of concepts and methods, further analyses on (1) the feasibility of CSI approach to other datasets and its links with others indicators (2) its sensitivity to different pressures (land uses changes), both accounting for variations across space and over time, highlighting its interest as an indicator of Biodiversity health at European scale

Introduction

Measuring the progress towards halting the loss of biodiversity is not an easy task as biodiversity cannot be evaluated as such. Indeed, biodiversity not only embraces species, ecological, and genetic diversity but also compositional, structural, and functional diversity as well as ecosystem functions and services. Ideally, the variables to be used must both describe biodiversity and show some reliable responsiveness to its changes. Furthermore, such variables need to be available at the needed scale – in the present case mainly at large scale, country or continental – as the exercise is not intended to be purely theoretical but at the contrary very practical. This task has already proven to be very complex: there have been many trials to assess biodiversity state, either using purely descriptive information (such as species or numbers of species) or functional variables (such as ecosystem services or resilience capacity). Currently, none has been entirely satisfactory, either because of conceptual problems (are species numbers really describing biodiversity status?) or because of feasibility (how to assess ecosystem services variations? Mace 2007).

In this report we will test the feasibility, sensitivity and reliability of a proposed assessment using commonly available datasets. **We propose to test a Community Specialisation Index (CSI) as a proxy to reflect the biotic Homogenization process and therefore the state and health of biodiversity and ecosystems**. Indeed, ecosystems under stress such as habitat loss, fragmentation or other disturbance including pollution suffer from a decline of specialist species while generalist species increase (Marvier, 2004). This process can be quantitatively estimated.

- In a first step, using literature references, we examined the conceptual evidence that Community specialisation index is a strong relevant indicator of biodiversity state and ecosystem integrity.

- Then, focusing of the species specialisation, we examined the relevancy of different calculation methods that account for preferences of species per habitat (Expert European database versus standardized collected data from monitoring) and assessed relationships between species specialisation and rarity/vulnerability indices (range, protection status) for species in Europe.
- We thus examined the relevancy of the Community Specialisation approach at large scale in Europe to address biodiversity trends under global changes, taking into account the characteristics of the available biodiversity datasets:
    - (i) We tested its sensitivity to land use changes using CSI from mammals, amphibians and reptiles atlases at spatial scales.
    - (ii) We addressed CSI temporal trends in Europe.
        - * We first examined the feasibility of using GBIF database to obtain temporal trends of biodiversity health at large scale (using bird model) Indeed standardized monitoring only occurs for a few taxonomic groups and only until recently at large scale. As the GBIF database is constructed with the aim to give access to available information on “overall” biodiversity by regrouping different national/regional databases all over the world, it is expected that it could offer the possibility of analysing the overall biodiversity trend in the past on long term series for many taxonomic groups at large scale.
        - * Second, using standardized collected data from bird monitoring, we examined CSI trend across the two last decades in Europe and explored the influence of possible drivers of CSI changes across time (Land cover changes, agriculture intensification and withdrawal).

Part I. The need for an indicator which reflects biodiversity state and ecosystem integrity

I.1. Different types of indicators

Indicators based on a Mean Species Abundance approach (MSA)

With the adoption, by heads of states, of the 2010 biodiversity target, a framework for indicator development has been agreed-upon, both at global, European and in some cases at national level. In Europe, the process on Streamlining European Biodiversity Indicators to assess progress towards the 2010 target (SEBI 2009) has resulted in the identification and documentation of 26 biodiversity-related indicators, clustered into six focal areas (European Environment Agency, 2007). Among them, in a pressure-state-response perspective, two focal areas especially focus on the biodiversity "health": "State and trends of components of biodiversity" and "Ecosystem integrity". We need to know how react biodiversity components across time facing global pressures and facing environmental policy responses adopted. We also need to know the state and trend of ecosystem integrity as it reflects ecosystem functioning that supports ecosystem services from whose human well-being depends. But while there is now a large consensus about the indicators of state and trends of components of biodiversity (SEBI 2009; Butchart et al. 2010), fewer have been proposed to reflect ecosystem integrity.

Ideally, a biodiversity indicator should accurately reflect changes in biodiversity, link such changes appropriately to specific pressures, and be rooted in sound scientific theory (Gregory et al. 2005: Annexe 1). Indicators based (i) on many species and (ii) on changes in species abundance are more prone to reflect biodiversity health than those based on few species and on presence-absence data. At the moment, existing indicators such as "Trends in the abundance and distribution of selected species" are among the most accurate indicators (see SEBI 2009; Gregory and van Strien, 2010; Butchart et al. 2010). They are based on species trends and for readability, trends are aggregated. So they can be grouped in the family of "Mean Species Abundance indicators"[1]. They correspond for example to the Living Planet Index (LPI) and its disaggregated associated indicators (Temperate living Planet Index, Terrestrial Living Planet Index…).

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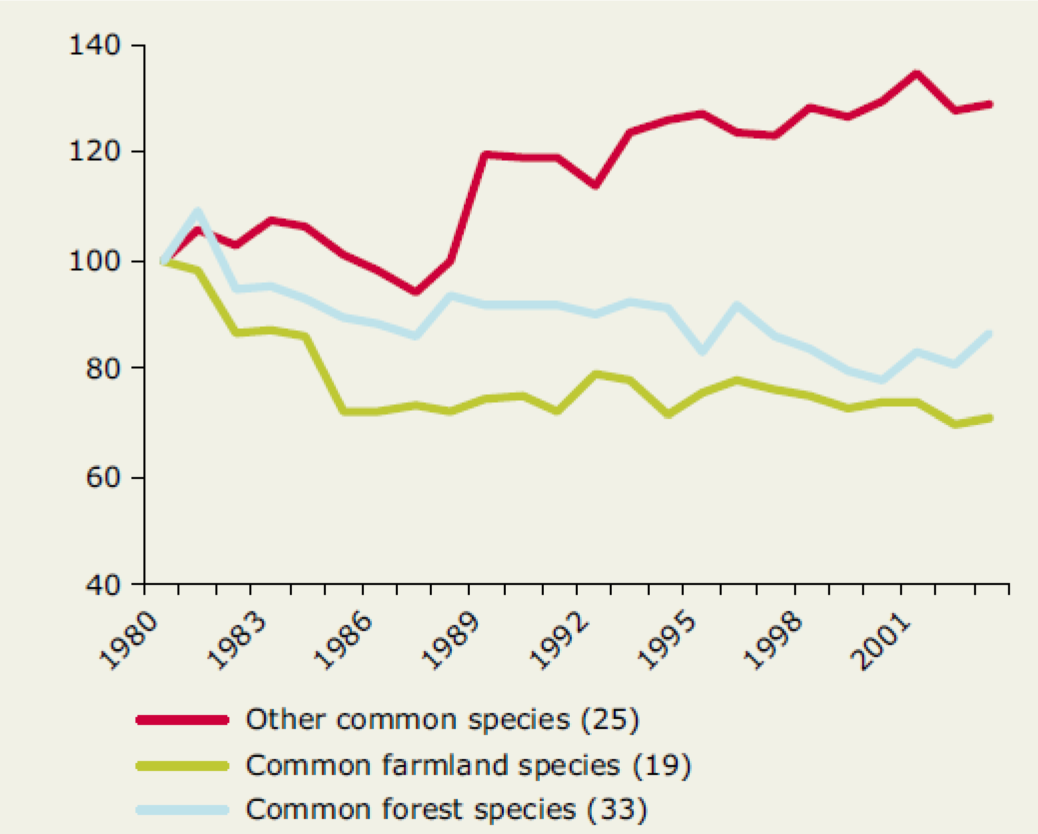
[1] The Mean Species Abundance (MSA) is an index which calculates the mean trend in population size of a representative cross section of the species (Netherlands Environmental Assessment Agency, 2006).
http://www.unep-wcmc.org/GLOBIO/PDF/Flyer_%20IB_Project_LR.pdf

However, accounting for all species could not very well reflect biodiversity health and ecosystem functioning as all species do not react to pressures in a same way and have not the same effect on ecosystem functioning.

Indicators based on a Mean Species Trait Abundance approach (MSTA)

Species responses and effects both depend on their traits (Hooper et al. 2005, Devictor et al. 2010). Thus, considering not only change in species abundance but also change in species trait abundance is expected to provide relevant information on mechanisms of Biodiversity changes. So “Mean Species Trait Abundance” indicators are complementary to “Mean Species Abundance” indicators (Devictor et al., submitted). For example, the Marine Trophic Index (MTI) gives supplementary information on ecosystem functioning compared to the Marine Living Planet Index, typical indicator of the MSA family. The MTI indicator is a functional indicator (MSTA) based on food-web theory, and has proven its usefulness in summarizing the impact of fisheries exploitation on marine ecosystems (Pauly et al. 1998), the most impacted species being those of the higher trophic levels.

The trait considered, and consequently the indicator proposed, can differ according to the pressure considered. The Marine Trophic Index, based on trophic level is expected to reflect the impact of fishing pressure on marine biodiversity, as fishing logically has more impact on species of higher trophic level. Different indicators should thus be considered on different groups defined according to the species trait value (size, diet, habitat preferences) allowing trend comparisons among groups. For example, the usually Common Bird Index refers to mean populations trends of common bird species in Europe according to their habitat preferences. It shows that farmland and forest birds strongly decline while generalist species that do not exhibit particular preferences for one habitat type increase (Fig. 1).



<u>Fig. 1:</u> Trends in abundance and distribution of selected species: Pan-European Common Bird Index as proxy for biodiversity health.

Green and blue decreasing lines respectively represent forest and farmland bird species' assemblage. Red line shows the trend of 25 other species showing no specific relation to any of the other main habitats i.e. species being distributed evenly among other habitats. Source: SEBI2010

However, we need indicators that can be applied generally i.e. that do not depend on particular species groups, but that reflect biodiversity state and ecosystem integrity facing the different pressure types. We thus need indicators which can reflect general causality relations with biodiversity health. In this context, focusing on species specialisation is shown to be a very interesting way to reach this goal as there is both conceptual and empirical support that biotic Homogenization is a consequence of biodiversity stress.

I.2. Indicators based on species specialisation as indicators of biodiversity state and ecosystem integrity

1.2.1. A major symptom of Biodiversity loss: decline of specialists species

A major symptom of biodiversity loss is indeed the strong decline of specialist species. This trend is observed all over the world in many taxonomic groups, including plants (Rooney et al. 2004), coral reef fish (Munday 2004), birds (Julliard et al. 2004, Devictor et al. 2008), mammals (Fisher et al. 2003), beetles (Desender et al. 2010), and butterflies (Ekroos et al. 2010). And paleontological studies based on fossil records suggested they seem to be more prone to go to extinct than generalists (see in Clavel et al. 2010). In contrast, generalists have been shown to benefit from moderate disturbances, being favoured under global changes, and being more likely to become successfully established when introduced (Clavel et al. 2010). Seven out of eight comparative analyses have for example shown a significant relationship between establishment success and ecological specialisation (in birds and fishes; Fisher and Owens 2004).

The most consistent feature of winning species is their large resource gradient, which allows them to survive everywhere. They are thus called **generalists**, while the others with affinity for specific conditions are called **specialists**. Ecological specialisation is then defined as the restricted ecological niche breadth (or width) of a given species (Futuyma and Moreno, 1988), specialists have a narrower niche width than generalists. This narrower niche width renders them more sensitive to changes.

Box 1: Specialisation in reference to the ecological niche of species

This degree of specialisation refers to the ecological niche of species. The ecological niche is defined as the overall (environmental) conditions in which a species can survive and have reproductive offsprings. Of course these environmental conditions are numerous, referring to several abiotic (temperature, humidity, light, salinity…) and biotic conditions (relationships, food resources…). When considering a gradient of one condition type, for example the temperature gradient, some species are able to develop in very different temperatures, i.e. across large amplitude of temperature conditions: they have thus a large temperature niche width and can be considered as thermal generalist species. In contrast, thermal specialist species tolerate a narrower gradient of temperature. Considering several environmental gradients, the niche can be illustrated by a larger "hypervolume" in space for generalist than for specialists (Fig. 2: Example of theoretical niche size of generalists and specialists for two environmental gradients).

A species i does not develop in a same way all along an environmental gradient, its fitness varies: indeed at the extremities of the environmental gradient that is tolerated by the species i, it will grow worse and will have less offsprings which will be in addition less numerous to survive: it "performs" worse (Fig 2, right).

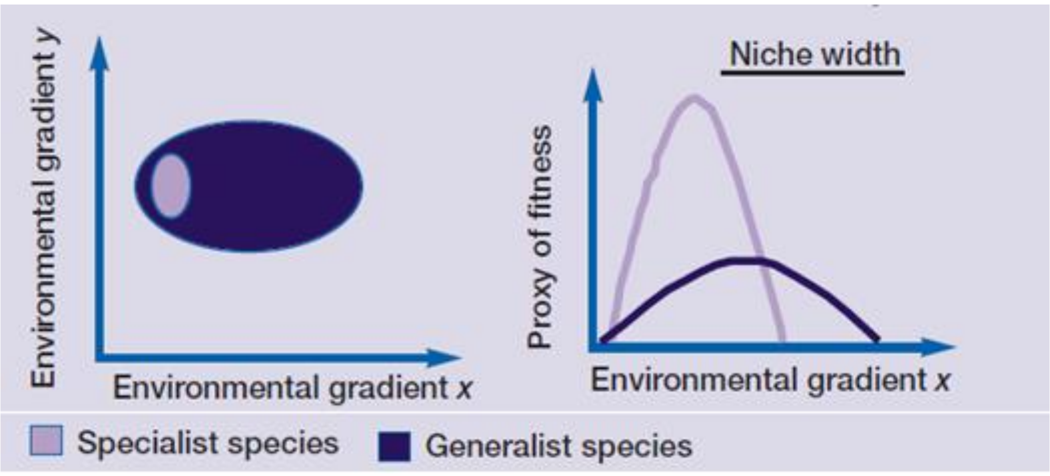


Fig. 2: Concept of the ecological niche and two different measures of it

In literature, the ecological niche is described with two aspects: on one hand, species response to a given set of variables considered as resources; on the other hand, species impact on the environment (see Devictor et al., 2010).

Here, with CSI, we focus on the first dimension, response of species to the resources as our aim is to benefit from indicator of Biodiversity state and integrity facing global changes. And indeed, the declines of specialist species are suggested to be related to disturbances to habitat and climate.

Box 2: Niche partitioning

Specialisation has to be viewed in an evolutionary perspective, accounting for the interactions among species. Theory predicts that in stable environmental conditions, species evolve to become more specialised so that the competition among species decreases through a better sharing of resources. This phenomenon can be interpreted in terms of niche partitioning among species (Fig. 3) and results in a higher specialisation of species, i.e. specialist species are more efficient at exploiting resources than generalist species. A well-known example of this is the beak shape of Darwin's finches in Galapagos.

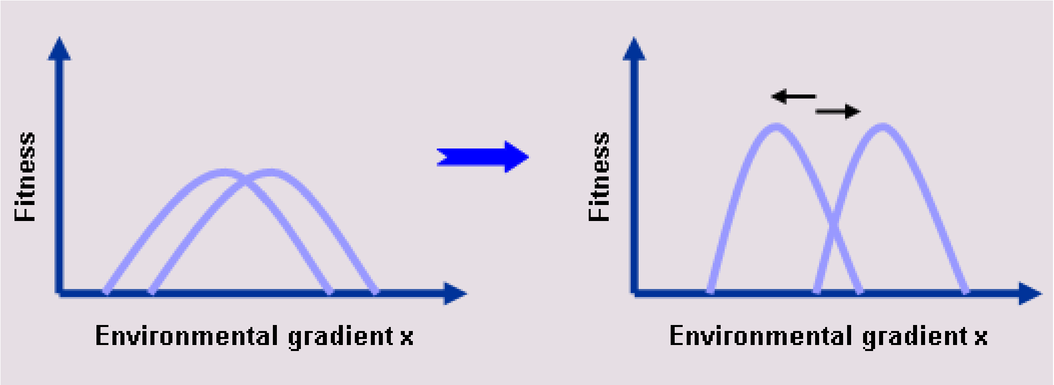


Fig. 3: Evolutionary principle of niche differentiation across time through selective pressures. The ecological niche of two species that coexist tend to differ across time driven by a process of competition reduction. On the right, the niche space shared by the two species is narrower than in the figure on the left.

As a consequence, **due to this differentiation of the species niches (and consequently of their functions), a community of specialist species provides a higher complementarity among species than a community of generalists.**

1.2.2. Biotic Homogenization process

Functional Homogenization

The both opposite trends of specialists (decrease) and generalists (increase) under global changes results in a higher similarity of biota among areas, both visible at taxonomic (TH) and functional levels (FH), that result in a Biotic Homogenization process (BH) (McKinney and Lockwood, 1999; Olden, 2006).

Indeed, most species are declining as a result of human activities ("losing species") and are being replaced by a much smaller number of species ("winning species"), that result in the replacement of local species by other, more widespread. In effect, this process “reshuffles” existing species distributions and reduces spatial diversity. Communities become more and more similar. Because generalist species have larger niche width (and thus are more widespread) they are more prone to be similar in a functional point of view than specialists (Box 1). So the taxonomic homogenization is accompanied by a functional homogenization that is highlighted by the decrease of the average specialization of community (CSI, Fig. 4).

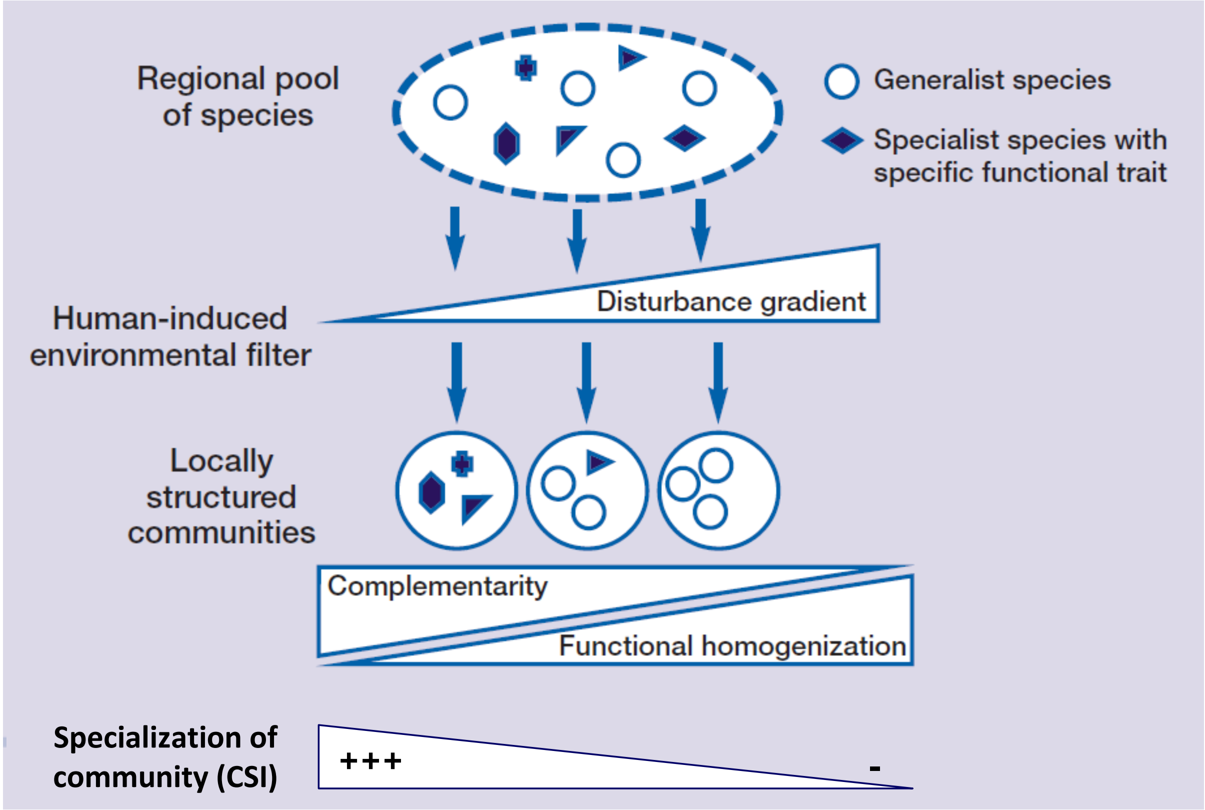


Fig. 4. Diagram describing how loss of specialists engenders loss of functional complementarity and thus functional Homogenization (from Clavel et al. 2010, partly modified)

Note that different mechanisms are proposed to explain the Biotic Homogenization process (Box 3)

Box 3: Mechanisms that may explain the Biotic Homogenization process
(Clavel et al., 2010)
Three mechanisms may dictate the outcome of global change in the balance between specialist and generalist species abundances (Clavel et al. 2010):

(1) Global changes may have direct negative effects on specialists, irrespective of the presence of generalists. For example, many European wetland species are declining because wetlands have been disappearing throughout Europe. Species adapted to that habitat are more affected because they cannot access the kinds of alternative resources that generalist species can.

(2) Because they are more flexible and innovative, generalist species may have the ability to colonize new niches that have been created as a result of global change.

(3) In many cases, global changes may have the same positive (or negative) effects on both specialists and generalists, but not to the same degree.

Competition induced by these differential responses determines the relative success of generalist species. For example, unusually warm spring seasons may favour the reproductive success of all species, but may be more favourable for generalists, due to their greater adaptability. Finally, combinations of these three mechanisms may further promote the success of generalists (Clavel et al. 2010).

The consequences of community changes on ecosystems

The phenomenon of Functional Homogenization raises numerous questions about the future of disturbed and transformed ecosystems on ecological and evolutionary time scales (Olden et al. 2006; Clavel et al. 2010). Species that are highly specialised are replaced by generalist species which share more similar functions, so the “newly” created community should perform less efficiently (Fig. 4). So the question is: **How do changes at the community level alter ecosystem functioning and ecosystem productivity, and do ecosystem services deteriorate in such circumstances?**

Some models show that functional characteristics of community (instead of diversity per se) strongly influence ecosystem properties (Díaz et al. 2007). Because generalist species may be considered as redundant to other species concerning functional characteristics, due to their niche width, some authors proposed that their diversity is not fundamental to maintain functions at the ecosystem level, as long as all functional groups (i.e. specialists) are present (Clavel et al., 2010). Of course, it is likely different concerning specialist species. Other models taking into account species traits, assume that presence of each species can help others to utilize resources differently (Tilman et al. 2001): some species are complementary in their patterns of niche occupation (for example: specialist species: see box 2, Fig. 3) and then can increase average rates of productivity (see Clavel et al., 2010 for details). Thanks to the complementarity among species within a community composed of specialists, the resources should be exploited more efficiently than in a community composed of generalists. And indeed, examples show that the ecosystem function (i.e. parasite regulation) is better performed by specialist communities than generalist communities (Finke and Snyder, 2008).

In a similar approach, a species-poor community composed of generalist species should lead to lower resistance (for example: resistance to introduction of invasive species) and lower resilience to changes than a community composed mostly of specialists (Clavel et al. 2010).

At a larger scale, when functional homogenization affects several communities, they become more similar and therefore they have more risks to react simultaneously in a same way facing a disturbance.

Consequently FH decreases the viability of the whole system, by decreasing the variability in the communities' responses to disturbance (Olden 2006, Clavel et al. 2010). Indeed, having a range of species that respond differently to environmental perturbation can stabilize ecosystem processes (Hooper et al. 2005), because the responses of species (and communities) differ. Although a given specialist species may be more negatively affected by disturbance than a generalist species, an entire (meta)community composed of many specialized species should be relatively less affected, on account of greater niche complementarity. Note however that under suboptimal or variable conditions, generalist species may also contribute to more efficient ecosystem functioning (Richmond et al. 2005).

Functional Homogenization: an indicator of biodiversity loss

As shown previously the replacement of specialist species by generalist species could have severe consequences on community and ecosystem functioning. Therefore, Functional Homogenization measured as the proportion of specialist species in the community is a good indicator because it measures the state of biodiversity, which is directly linked to drivers of global changes.

There is thus strong conceptual evidence that Community Specialisation, reflecting the relative composition between specialist species with a narrow ecological niche and generalist species which are less requiring, **can be a good indicator of biodiversity state and ecosystem integrity facing global changes**. However it must be mentioned that this measurement being an average of the community specialisation, high CSI values are expected for both rich and poor communities composed of specialists. High CSI values could be expected in extreme environmental conditions, where only few species having particular adaptations could maintain. These extreme conditions occur in particular natural habitats but may also characterize high degraded habitats such as for example very high intensive agricultural areas. In these specific cases a double approach including the CSI and species richness can be useful

Part II. Quantify specialisation: state and trends facing global changes

To build the Community Specialisation Index, we first need to assess the specialisation degree of each individual species in the given community, i.e. to assess the **Species Specialisation Index** (**SSI**). Then, a **Community Specialisation Index** (CSI) can be calculated for each study area as average of species Specialisation Indices of all species being present in the study area.

If previous studies have shown the relevance of this approach, especially at European scale (Loïs et al. 2009), some points need further investigations.

- How to quantify the specialisation of each species?
- the sensitivity of CSI to different pressures

II.1. Species specialisation: quantifying, relationships with other indices, drivers

II.1.1. Quantifying the Species specialisation index

The need for further investigations about the species specialisation measures

There are different ways to measure species specialisation (Devictor et al. 2010). A clarification of specialisation quantifying methods have been proposed (Devictor et al. 2010), However difficulties remain when measuring specialisation due to the general lack of data available at large scale. Here, specialisation of a species should represent the **variance in species' performance across a range of environmental conditions**, broadly defined by one or several biotic (and ⁄ or abiotic) resources. So to measure an index of specialisation per species we need to define a relevant environmental gradient along which to measure niche width and a proxy of species' performance for the values of this environmental gradient.

(i) The specialisation to habitats has been proposed as a good proxy of the overall environmental conditions. A specialist species is thus defined as a species that have distinctive "preferences" for a few habitats of the overall gradient, while a generalist "occurs" in a higher number of habitats.

(ii) Species performance across the range of habitats is meanwhile proposed to be addressed as species abundance or presence per habitat class (Julliard et al. 2006).

Indeed, a major constraint in applying these indicators is the availability of data. Of course, the more quantitative data are, the more biologically meaningful are the calculated indices. However, standardized collected data on species abundance highlighting their performance across the range of habitats generally lack for many taxonomic groups, and especially at large scale.

In a previous report on CSI, Loïs et al (2009) tested different ways of estimating the Species Specialisation Index (SSI) according to the type of available data (Annexe 2): using quantitative data from extensive monitoring programmes on one hand and presence-absence data based on expert knowledge on the other hand. Considering species specialisation based on habitat preferences, the SSI can be calculated:

(1) as the variation coefficient of the abundance of a species across the habitat types (quantitative data available) or

(2) as the proportion of habitat types used by species (expert knowledge). Then, they showed strong correlation between SSI calculated from presence-absence data or abundance per habitat using as example the French breeding bird survey.

However, many questions remain on how to quantify specialisation in practice, including:

- about the classes of habitat to consider: Which classification? How many classes to be considered? Working on a large scale such as the European scale, how to take into account the lack of particular habitats in some geographic regions?
- about the relevancy of using expert knowledge to estimate species "performance".

In addition, we need relevant measures of species specialisation at large spatial scale. The SSI calculated in France from French Breeding Bird Survey may not be systematically valid for all species in other regions. Indeed, the specialisation of a species may partly vary across its range size. Moreover, some of the overall habitat classes defined as the range of habitats in Europe may not occur in the spatial geographic range of a species, which can lead to the species being wrongly qualified as more specialized than others due to the method of calculation.

We thus need to better explore the correlations among indices calculated from different sources (expert, monitoring) using different habitat classes to better know the relevance of measuring specialisation for other taxonomic groups from expert knowledge.

Assessing the relevance of expert knowledge dataset to calculate SSI: using French Breeding bird survey and Eunis dataset

First, we examined the relationships between a Bird species SSI calculated from Expert knowledge (using EUNIS dataset) and from standardized monitoring data (French breeding bird survey), and secondly, we examined different methods of calculation taking into account the number of habitat classes and their geographical range.

Methods

We used the Bird EUNIS database (available at the ETC-BD, see Van Kleunen, 2003), which describes species affinity for nesting and foraging per EUNIS habitat (n = 98, EUNIS level 2 classification) for 493 European breeding bird species, per geographic region (11). Levels of affinity (primary habitats, secondary, others) were coded from 3 to 1. We used the higher level of affinity per species-habitat pair and calculated SSI as the variation coefficient of species affinity across the range of habitats.

We calculated SSI per geographic region (SSI_rgeo) and SSI across habitats that are only present in the geographical regions where species occur (SSI_p) for the 493 species. We also calculated SSI values across all habitat classes (SSI_all, n= 98, n = number of overall habitats classes available in the database) and across a reduced number of habitat classes, i.e. 10 groups (SSI_EUNIS-grouped habitats: 10 classes of "similar" habitats corresponding to the level 1 of EUNIS classification).

Then we examined correlations between these indices and those derived from monitoring within the French Breeding Bird Survey (see Loïs et al. 2009, using abundance across a range of 18 habitat types, n species =176) and the SSI_EUNIS calculated as the average of SSI_rgeo for the France (4 regions).

Results

Significant correlations were found among SSI calculated from the two methods:

- from a different number of habitat classes (Fig. 5), SSI calculated from 98 versus 10 habitat classes are strongly correlated.

- from standardized collected abundance (monitoring schemes) versus from expert knowledge (EUNIS database) (Fig. 6).

These results show that:

- **Although considering a higher number of habitat results in a more precise SSI calculation and then to a higher sensitivity of the further calculated CSI, the SSI can also be accurately calculated from a small number of habitats when only this data type is available.**
- **SSI is conservative and expert knowledge database can be used to calculate a relative specialisation index of species thanks to strong correlations among SSI calculated from different sources. This result is confirmed by other studies (Reif et al., 2010). Note that SSI can be also calculated using other expert databases even if EUNIS databases should be preferred. For example, we also found correlations for birds among SSI_monitoring and SSI calculated from the BioScore database (http://www.bioscore.eu/) which gives expert information on the species sensitivity to changes per habitat (Corine Land cover habitat classification). Correlations were also found among SSI_Eunis and SSI_BioScore for amphibians and mammals.**

Next, when using bird SSI values at large scale, we used “SSI_p “ (Annexe 3), i.e. SSI calculated from EUNIS database accounting for the habitats of the geographic regions where species occurs (see p 18).

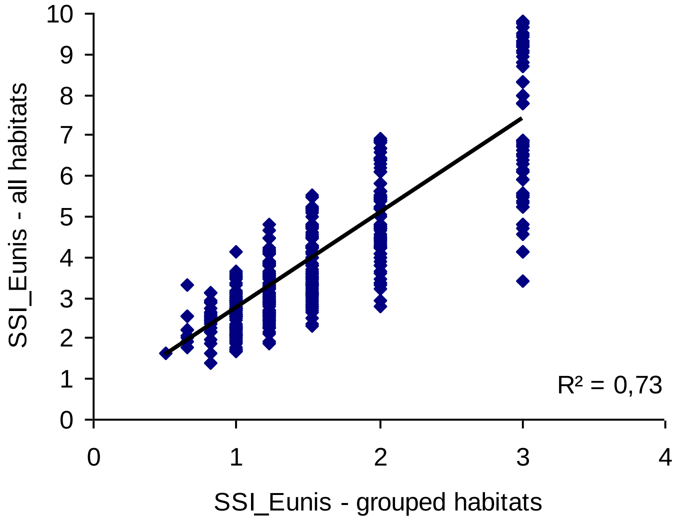


Fig. 5. Evidence of strong correlations among SSI calculated from different numbers of habitat classes. (n species = 493, p < 0.001, SSI_Eunis – grouped habitats : SSI calculated from the 10 higher class of Eunis habitats (level 1) and SSI_Eunis_ all habitats calculated from the 98 habitats classes available (level 2 and 3),. Here, SSI were calculated using the following formula described in Loïs et al. (2009). Note that a strong correlation occurs among SSI calculated using variation coefficient and the formula (R² = 0.93)).

$$SSI = \left(\frac{H}{h} - 1\right)^{\frac{1}{2}}$$

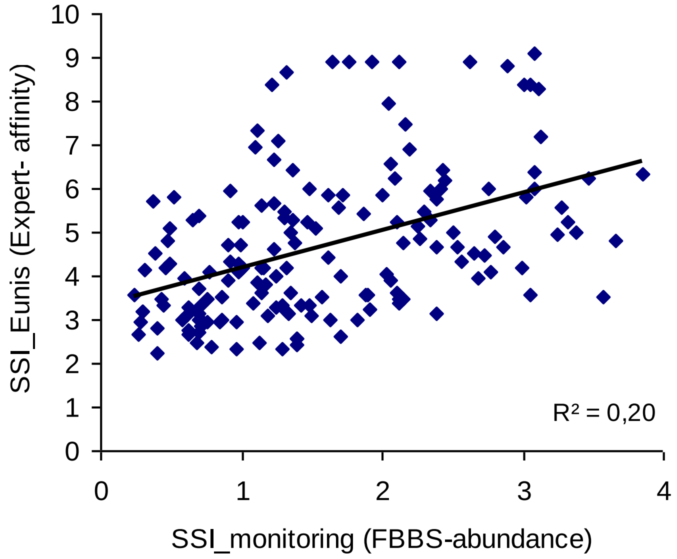


Fig. 6. Correlation between SSI calculated from different type of monitoring versus expert data. (n species = 176, p < 0.001, SSI_monitoring was calculated as the variation coefficient of species abundance across the 18 habitat classes using the French breeding Bird survey, SSI_Eunis was calculated as the average SSI of SSI_rgeo of 4 regions in France (atlantic, alpine, continental, Mediterranean) using Eunis database (expert knowledge, 98 habitat classes).

II.1.2. Relationship between SSI and rarity (range and “protection status”)

Niche breath partly determines the rarity of species. The rarity of a species is defined according to three linked dimensions: its abundance, its geographical range, its niche breath (ecological exigencies) (Annexe 4).

i) Thus the SSI is expected to be correlated to rarity, i.e. the area of species geographical range.

ii) We could also expect a positive correlation between SSI and vulnerability status (IUCN) of species and with “protection status” because the rarity partly influences the extinction risk of species. On the other hand, generalist species could be vulnerable due to human pressures such as hunting not directly linked with niche width. And species of a restricted area could be considered as vulnerable while they could be generalists in their distributional area.

Therefore, we examined i) relationships among SSI and the geographical range of species, and ii) among SSI and vulnerability status and protection status for birds, mammals, reptiles and amphibians at European scale.

Methods

We used SSI of mammals, amphibians, reptiles from Loïs et al (2009), and SSI of birds obtained as described before (SSI_p).

i) The area of geographical range was calculated from European Atlases (from the Atlas of Amphibians and Reptiles in Europe: Gasc et al. 1997, from The atlas of European mammals (Mitchell-Jones et al. 1999) and from The EBCC Atlas of European Breeding birds; Their distribution and abundance (EBCC, 1997) as the number of occupied cells (grid of square of 50 km x 50 km area). Pearson correlation tests were performed. Relationships were examined using linear models.

ii) Vulnerability status and protection status were respectively obtained using European IUCN red list and habitat directive (Annexe II of Habitat directive). Relationships were examined using linear models for mammals, reptiles, amphibians: We used binomial family for protection status, and converted vulnerability status in numerical values ranging from 1 (Least concern) to 5 (Critically endangered).

Results

i) Strong correlations occur between SSI and the area of **geographical range** of species (Fig. 7). Specialists occur in significantly smaller areas than generalists as shown for amphibians, mammals, reptiles and birds (Fig. 7). These observed relationships are not surprising as species exhibiting smaller geographical range meet a lower number of habitats (see Loïs et al.2009, for calculation method of SSI for mammals, amphibians and reptiles).

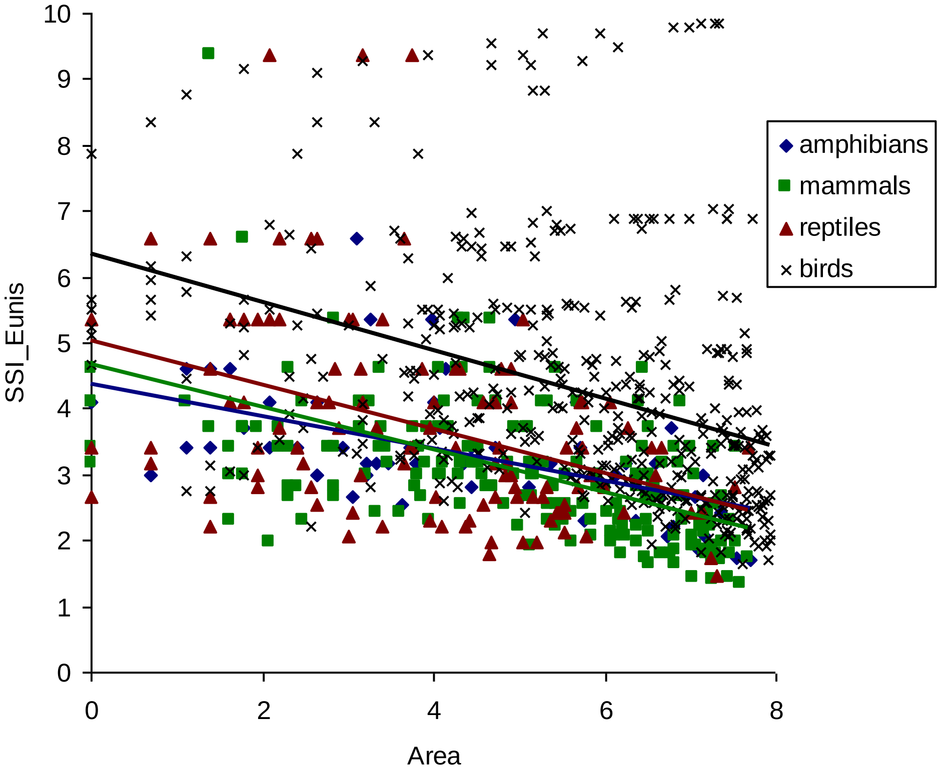
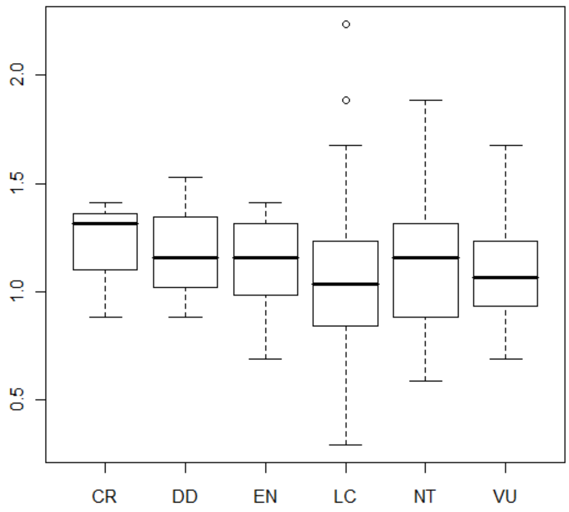


__Fig. 7__. **Relationship between SSI and size of geographical range for the four taxonomic groups**. The size of geographical range was calculated as the number of occupied cells of atlas grid, these data were log transformed. For the four taxonomic group, specialisation is linked with a narrower distributional range size: mammals: $t = -8.7465$, $df = 188$, $p < 0.001$, $R^2 = 0{,}29$; reptiles : $t = -4.5311$, $df = 113$, $p < 0.001$, $R^2 = 0{,}15$, amphibians: $t = -4.836$, $df = 59$, $p < 0.001$, $R^2 = 0.28$; birds :$t = -8.4781$, $df = 427$, $p < 0.001$, $R^2 = 0{,}14$)

ii) In contrast, we did not find significant relationships between SSI and **vulnerability status (IUCN red list)** (Fig. 8) for mammals, amphibians and reptiles. There is also no significant relationships between **SSI and protection status** (Annex II of Habitat directive)**,** excepted a weak correlation for mammals ($F_{1,189} = 218$, $p = 0.01$) and a negative correlation for reptiles ($F_{1,113} = 131$, $p = 0.005$). Thus, although niche width partly determines the rarity of a species, it does not mean that species recognized at risk are only specialist species. Indeed, the vulnerability and the protection status of a species are defined according to many different criteria (see IUCN and Habitat directive for details).



<u>Fig. 8</u>. Relationship between SSI and vulnerability status of mammals, amphibians, reptiles. Vulnerability status is along axis X (LC: Least Concern, NT: Near threatened, VU: Vulnerable, EN: Endangered, CR: Critically Endangered / DD –lack of data- were replaced by "NA" in the analyses, n= 311) and SSI (log transformed) is along Y axis.

So specialist species present narrower geographical range than generalist species. However, they are not more concerned by vulnerability and protection status than generalists. So SSI is not redundant with vulnerability and protection status, and then gives complementary information on species.

The main and important conclusions are:

Expert knowledge on species occurrence and preferred habitat-types can be used with a confidence, as compared to intensive detailed monitoring. Of course, when the choice is offered, using abundance collected from standardized monitoring should be preferred, but using expert knowledge database can still be used. This gives more opportunity for building an indicator on CSI on a European scale, taking into account that no comprehensive species monitoring is in place at continental scale for many different taxonomic groups.

Of course, it is thus of interest to develop such database of habitat preferences per species for other taxonomic groups. However, as specialisation of a species can differ with the localisation within its distribution range area, a better assessment of the relationship between species and habitats at finer scale (per region) should be helpful to gain sensitivity of the indicator.

Specialist species are rarer than generalist species in the point of view of geographical range but they are not more "covered" by vulnerability (IUCN red list) and protection status (Habitat directive) than generalist species. So specialisation gives an interesting complementary information.

II.2. Community specialisation index: trends and sensitivy to pressures

Many evidences support the view of the relevance of using the CSI as indicator of biodiversity health at large scale and, consequently, at the European scale. First, there is strong conceptual evidence about the relevancy of its use as indicator of biodiversity state and ecosystem integrity (see before and Clavel et al. 2010; Devictor et al. 2010). There is also empirical evidence of its sensitivity to global changes at both local and national scales (Devictor et al. 2008; Kerbiriou et al. 2009; Fillippi-Codacioni et al. 2010; Doxa et al. 2010) when applied on bird data (abundance, presence-absence). And, Loïs et al (2009) have shown the feasibility of applying the approach at a European scale using presence-absence data of Atlases.

Taking into account the constraints linked to the availability of data, we need to evaluate its sensitivity across time and its sensitivity to pressures at large spatial scale.

- As in many cases, only atlases data are available, we tested the CSI sensitivity using such data (CSI calculated by Loïs et al. 2009 for mammals, reptiles, amphibians) facing different land use pressures.
- We also explored the possibility of using GBIF dataset that regroup data for many different taxonomic groups at large scale across time. The possibility of using such GBIF data was tested by comparing the relevance of patterns found from GBIF data in comparison to known biodiversity patterns (Atlas).
- Finally, using standardized collected data (from standardized bird monitoring) we examined CSI trend across the two last decades in Europe for birds. Then we addressed possible drivers of CSI changes across time.

II.2.1. Sensitivity of CSI to spatial change: Relations between spatial distribution and variables of anthropogenic intensity

In order to check whether the proposed indicator was reactive to biodiversity health we examined correlations among CSI from European atlases (see Loïs et al. 2009) with variables that are known to be anthropogenic and of suspected pressure upon biodiversity. We compared the CSI responses to responses of other biodiversity indices (richness) to these pressures. Of course, we took into account the effects of geo-climatic variables and spatial autocorrelation.

Methods

We used CSI and richness of mammals, amphibians and reptiles calculated by Loïs et al (2009) from European atlases (Gasc et al. 1997,– Mitchell-Jones et al. 1999). We crossed them with different sets of variables describing land use pressures and calculated per each cell of the atlas grid (50 km x 50 km).

Thus we used:

1) Variables describing fragmentation, urbanization and agricultural intensification in Europe proposed by ETC-LUSI[2]

2) Variables describing land-use cover and habitat diversity. Using CORINE Land Cover dataset (2000, level 3), we calculated for each cell of the grid (50 km x 50 km) the overall area of each habitat class (next combined in natural, urban, agricultural), the average size of polygons per habitat, and then an index of habitat diversity (Shannon diversity Index, that account for the number of polygons and the area of polygons). Pressures data were not available for all cells of atlases, what explains the decrease of considered cell number in comparison with Loïs et al. (2009) data.

Species specialisation can be driven by geo-climatic conditions that also influence the distribution of human density and thus anthropogenic pressures. In order to clearly distinguish these effects, we took into account geo-climatic conditions as co-variables. For the purpose we calculated for each cell of the grid the average altitude, and the average value of eight climatic conditions (annual mean temperature, temperature seasonality, maximum temperature of warmest month, minimum temperature of coldest month, temperature annual range, annual precipitation, precipitation seasonality, precipitation of driest quarter, precipitation of coldest quarter) using data layers from world clim site [3]. These geo-climatic conditions are correlated. So we grouped variables using clustering methods according to correlations and then we retained one variable of each cluster group in the model (break point = 0.9). We selected these co-variables among the others of each group as those that have the stronger effect on CSI. So, for the three taxonomic groups, we retained as co-variables minimal temperature of coldest month, annual precipitation (log transformed) and precipitation seasonality (log transformed), and the altitude for reptiles and mammals. As there is an obvious geographical factor regarding CSI layers, very likely due to recent evolution history since the last ice age period, we also integrated in our models latitude and longitude.

In a second step, we used a subset of data to examine the effect of land use pressures, with “reducing” the effect of geoclimatic variables in analyses. To limit the influence of altitude on CSI of mammals, we examined the influence of land use pressures where altitude is less than 150 meters (n=435 cells).

For overall tests, we performed generalized least squares models (gls, using geographic coordinates) to take into account spatial autocorrelation. So the effect of the pressure variable on CSI was tested adjusted to geo-climatic co-variables and with accounting for spatial autocorrelation.

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[2] http://www.worldclim.org

[3] Lnmeff : Fragmentation of landscape by constructions, roads and railways, C1: Smoothed artificial areas which corresponds to a spatial aggregation of all CORINE LC classes included in the ¨Artificial surfaces¨ category – CLC level 1, class 1, C2: Smoothed intensive agriculture areas, which corresponds to a spatial aggregation of Arable Land , Permanent crops and Annual crops associated with permanent crops – C1 and C2: CORILIS methodology

Results

Our results confirmed those of Loïs et al. (2009). All variables taken together do explain massively CSI variance. As expected given the history of the south-north continental colonisation and species evolution, geographic coordinates (see Loïs et al, 2009) and geo-climatic variables do explain the biggest part of variance.

- About the influence of the geoclimatic conditions on CSI[4],

- the CSI of mammals was explained positively by altitude, by annual precipitation and negatively by precipitation seasonality The effect of these variables was tested adjusted to geographical coordinates and to the other geoclimatic variables
- The CSI of reptiles was first of all driven negatively by precipitation seasonality and altitude and only in a fewer part by annual precipitation
- The CSI of amphibians was driven positively by annual precipitation and only in few parts negatively by precipitation seasonality and positively by the minimal temperature of the coldest month).

Overall Europe: Effect of land use pressures

- We tested also the effect of land use pressures (see before) on the CSI taking into account the geographic and geoclimatic variables.

- Not surprisingly given the previous result, we found **no significant negative trends for the land use pressures** (fragmentation, urbanization and agricultural intensification).
- However, we found a significant **positive effect of natural habitat[5]** (per cell of 50 km x 50 km, calculated from the different manner: natural habitat Shannon diversity index, number of natural habitats polygon per cell, size of natural habitat polygons) on the CSI of the three taxonomic groups amphibians, reptiles, mammals.

Europe below altitude of 150 m: Effect of land use pressures

- Interestingly, CSI was found **sensitive to land use pressures** when subsets of data that reduce the effect of geoclimatic variables in analyses were used. Here, the influence of altitude on CSI of mammals was reduced using a data subset where altitude is less than 150 meters (n=435 cells).

- We thus found a **significant negative response of CSI mammals to fragmentation** by roads, railways, constructions while a positive response of richness was found (Fig. 9). Other consistent expected trends of CSI and richness were found according to the increase of **urbanisation** and **natural area** per cell (Annexe 5).

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[4] Model: CSI~Lat+Lon+Lat^2+Lon^2+LatLon+geoclimvar1+ geoclimvar2+geoclimvar3+ geoclimvar4. CSI of mammals positively explained by altitude ($F_{1,1358}$=,24.37, $p < 0.001$), by annual precipitation ($F_{1,1358}=26.69$, $p < 0.001$, a quadratic effect occurs) and negatively by precipitation seasonality ($F_{1,1358}=18.66$, $p < 0.001$, a quadratic effect occurs). CSI of reptiles driven negatively by precipitation seasonality ($F_{1,934}=112.15$, $p < 0.001$) and altitude ($F_{1,934}$==17.48, $p < 0.001$), by annual precipitation (negatively $F_{1,934}=10.71$, $p = 0.001$). CSI of amphibians driven positively by annual precipitation ($F_{1,1223}=31.45$, $p < 0.001$), negatively by precipitation seasonality ($F_{1,1223}=6.56$, $p = 0.01$) and positively by the minimal temperature of the coldest month ($F_{1,1223}=4.54$, $p = 0.03$).

[5] Positive effect of natural area on the CSI of the three taxonomic groups: p=0.004

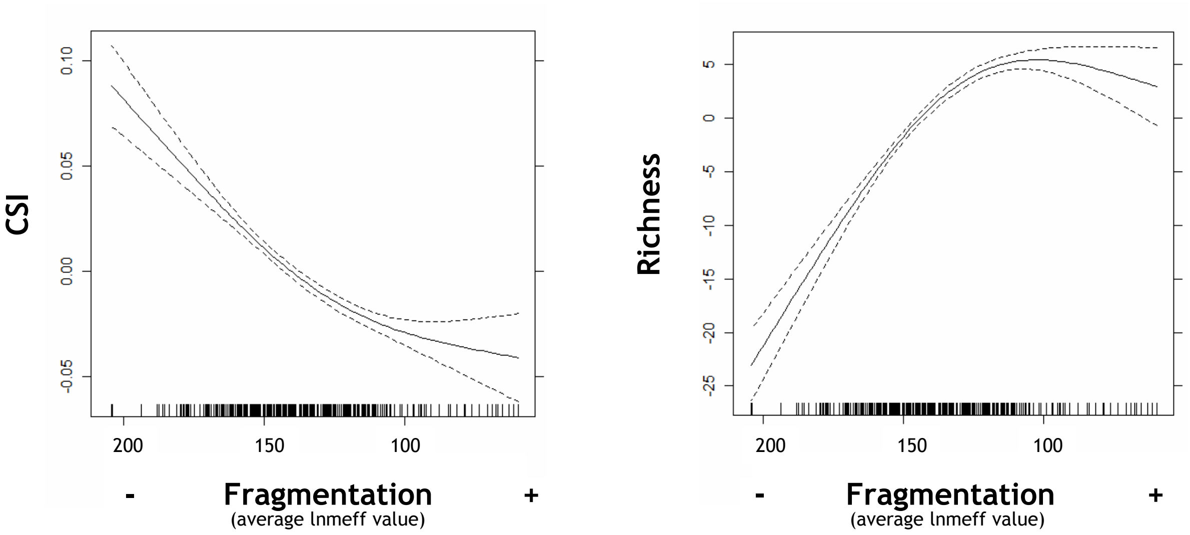

Fig. 9. **CSI response to anthropogenic pressures: the impact of the land cover fragmentation on the community specialisation of mammals in Europe** (altitude < 150 meters).
Fragmentation corresponds to the "lnmeff "variable provided by ETC-LUSI. This variable describes the land cover fragmentation by roads, railways, and urbanisation (using Corine land cover database), the higher the lnmeff value is, and the lower the fragmentation is. Average lnmeff value was calculated per cell of the atlas grid (50 km x 50 km). Generalized least squares models were performed to account for spatial autocorrelation. Then, the effect of the variable "lnmeff" on CSI was tested adjusted to geo-climatic variables and accounting for geographic coordinates. As mammal specialisation is mainly explained by altitude, we used a subset of data where altitude is less than 150 meters to test the effect of anthropogenic pressures on CSI. The CSI significantly decreases (p=0.028) when fragmentation increases, while in contrast richness increases when fragmentation increases (p=0.001).

So, our results confirm the interest of using CSI as an indicator of biodiversity health. Indeed they confirm its sensitivity and its meaningful trend facing global changes, highlighting the ongoing biotic Homogenization process. Of course, at large scale, CSI is first of all driven by geoclimatic variables (due to historical constraints and evolution). Nevertheless, using subsets, it is possible to show its sensitivity facing pressures although the CSI was measured from data of "poor sensitivity" such as presence-absence of species in cell of 2500 km² (Atlases data, size of cell: 50 km x 50 km).

Our results thus show that there are some very strong relationships between pressures upon biodiversity and CSI for taxa group of which distribution is not massively driven by biogeographical and/or thermal factors. **They confirm previous studies that showed at finer scale how much specialisation is correlated with urbanization, fragmentation and disturbance of landscapes and is even much more sensible than other species indicators, including species trends or richness, making it a good proxy of biodiversity health response to potential 'diseases'**. (Devictor et al. 2007, 2008a, 2008b).

II.2.2. The feasibility of using CSI trend across time in Europe as a result of biodiversity changes under anthropogenic pressures.

The relevance of an indicator depends of course of its capabilities to allow detecting biodiversity changes across time facing global pressures. So next we addressed CSI trend across time.

First, we explored the possibility of using GBIF dataset [6] that regroup data for many different taxonomic groups at large scale across long time series. Standardized monitoring only occurs for a few taxonomic groups and only until recently at large scale. As the GBIF database is constructed with the aim to give access to available information on “overall” biodiversity by regrouping different national/regional databases all over the world, it is expected that it could offer the possibility of analysing the overall biodiversity trend in the past on long term series for many taxonomic groups at large scale.

Second, using standardized collected data from bird monitoring, we examined CSI trend across the two last decades in Europe and addressed possible drivers of CSI changes across time (Land cover changes, agriculture intensification and withdrawal). In these two cases, birds were chosen as biological model thanks to the availability of different datasets and known trends that allow comparisons.

Assessing the feasibility of using GBIF data to address CSI trends across time

One of the main constraints in surveying biodiversity trend across time is the lack of data collected in long time series at large scale. In addition, when data are available the effort of collection is often unknown, therefore the comparison of indices such richness, diversity does not reflect biodiversity trends but rather the sampling effort.

Thanks to the method of calculation, the CSI could offer the possibility of assessing the biodiversity state across time using such not standardized collected data based on the hypothesis of a similar effort of sampling of specialists and generalist across time With the increase in sampling effort through time, more and more data on common species (probably more generalists species), have been collected, but on the other hand volunteers have also collected more and more information on rare species (often specialists), so that we could be confident in this hypothesis.

Of course, analysing and correcting the potential bias in such datasets is necessary before applying the CSI approach. Here we examined the possibility of using the GBIF database to analyse the CSI trend in Europe across time., GBIF database is a world database constructed with the aim to provide information at large spatial and temporal scale

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[6] http://www.gbif.org, Work in project with S. Pavoine (MNHN), C. Barbot

on biodiversity (), The presented work partly corresponds to a master degree submitted by Cyrielle Barbot (Barbot, 2010) in the University of Paris in June 2010.

Method

The possibility of using such GBIF data was tested by comparing the relevance of patterns found from GBIF data in comparison to known biodiversity patterns (Atlas).
As much information (through literature) and datasets allowing comparisons are available for birds, analyses were performed on this taxonomic group. We used an extracted subset of the GBIF database of birds in Europe (27 countries) that overall contains more than 25 000 000 data (in January 2010) from whose 6 500 000 are geo-referenced and dated records. We retained records relative to breeding terrestrial birds (i.e. records from April to July, non-aquatic birds) located in continental Europe (records located in the sea were suppressed), that resulted in 1 337 569 records (i.e. 5% of the initially proposed dataset) belonging to 281 species. These 1 337 569 records were distributed among 24 032 geographic coordinates mainly distributed among four countries (95% of data) from 1856 to 2009. Thus, the sampling effort differs through time across countries. As expected, the number of recorded species strongly increased with the increase of sampling effort across time especially from 1955.

With the aim to compare patterns, we grouped these records by time period (year, decades, period of fifty years) and by location using records projection on a grid of Europe. The size of the grid cell was chosen to be comparable to those of European Bird Atlas (50 km x 50 km, 1980-2000). We verified the absence of outlier data examining the distribution of record frequency per geographical coordinates and the concordance of geographical distribution of each species between Atlas and GBIF sources. Then, we identified cells that are “well” sampled using a comparison of observed richness and estimated richness per cell, calculated from the collected effort per species (see for details Barbot, 2010). According to these results, 36 % of the GBIF cells were enough sampled.

Then, we compared spatial and temporal patterns from the GBIF to known patterns (literature, Atlas), using species richness distribution across Europe and bird CSI distribution calculated from Atlas (ECCB, 1997) and SSI_p (Annexe 6, see p 18), distribution of Community temperature index (CTI) across latitude gradient (Devictor et al. 2008), temporal trend of the proportion of farmland birds, and temporal trend of Community temperature index (Devictor et al. submitted). Similarly to CSI approach, CTI is calculated for one community as the average of Species Temperature Index (STI), which is the average temperature experienced by individuals of one species over its distributional range (see Devictor et al. 2008). Finally, we calculated CSI per cell per decade for the best sampled period (1950-2009) for cells that were at least well sampled during three decades of the period. We then examined CSI trend across time using linear models.

Results

Our results showed that the use of GBIF database allows meeting acknowledged spatial patterns such as richness and CTI patterns. However, at this time, with the available GBIF data in 2009, we did not find known temporal trends, nor a significant decrease of the proportion of farmland species, nor an increase of CTI while it is the case with data of better quality such as abundance data coming from standardized monitoring (see Gregory and van Strien, 2010; Devictor et al. in prep). So not surprisingly, we did not find a decrease of the Bird CSI using the GBIF database (Fig. 10).

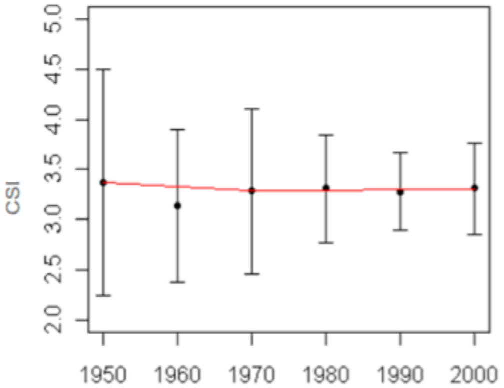


Fig. 10. The use of GBIF database does not allow us highlighting CSI trend for birds in Europe during the five last decades such as it does not allow us at this time finding known temporal trends obtained using standardized collected data (see for details: Barbot, 2010).

This work emphasizes the importance of cleaning and treating data (only 5% of GBIF data were finally useful) and confirms the spatial and temporal biases in sampling efforts. Quality controls about the data themselves (like research of outliers and accuracy of species distribution) are giving good results. Analyses of GBIF results in comparison with known trends tend to show that GBIF data can be used to study spatial structures (like species richness distribution or latitude gradient of CTI).

But they are of poor help to describe temporal trends (like farmland bird decline or change in functional community trait). Since the lack of information especially concerns temporal trends, these results tend to limit the usability of current GBIF data to highlight temporal trends of biodiversity under global changes through CSI approach. In the future, we can expect a better usability of the GBIF data with the increase of GBIF records. Note that even if the database seems to be large, b in reality, it does not contain at this time the main part of available databases on Biodiversity. For example, the used bird subset of GBIF only contains 77 000 records for the France between 2000 and 2009 (presence) while the STOC (French breeding bird survey database) contains more than 2 000 000 data for the same period (standardized collected abundance data).

Temporal trends of biodiversity are logically better addressed using standardized monitoring programs that are developed to this aim (while this is not the purpose of GBIF). We next examine CSI trends using such standardized collected data.

Trend of CSI across time in Europe using data from standardized monitoring

We examined CSI trend across time in Europe using trends of bird abundance data collected from standardized national/regional monitoring programs from five European countries.

This was developed in the context of a volunteer partnership project among scientific teams in Europe (Project with I. Le Viol, V. Devictor, L. Brotons, A. Lindström, J. Reif, C. Van Turnhout, F. Jiguet , and possibly others in the future - work not ever published).

Methods

We used a bird database joining the data collected by standardized monitoring national/regional programs from five countries: Sweden, Netherlands, Czech Republic, Catalonia and France.

The database contains more than 30 000 records, corresponding to more than 7 200 000 census birds, belonging to 234 species surveyed in more than 5500 sites during the two last decades (from 1990 to 2008).

Using SSI_p measured from Eunis database (see before), we calculated a CSI for each site for each year (average specialisation of the assemblage of birds detected per site per year) as such the SSI of each species is weighted by its abundance).

We examined CSI trend across time for overall Europe and by country using linear mixed models. In the overall model, the effect of year on CSI was tested adjusted to country effect and weighted by a sampling effort per country.

Results

CSI strongly decreases across the two last decades in Europe highlighting a strong functional biotic Homogenization process (Fig. 11). The overall trend is a decline during the overall period even if some fluctuations occurred: CSI declined during 1990-1993, 1997-2001, 2006-2008 periods and increase during 1993-1997 and 2001-2006 periods.

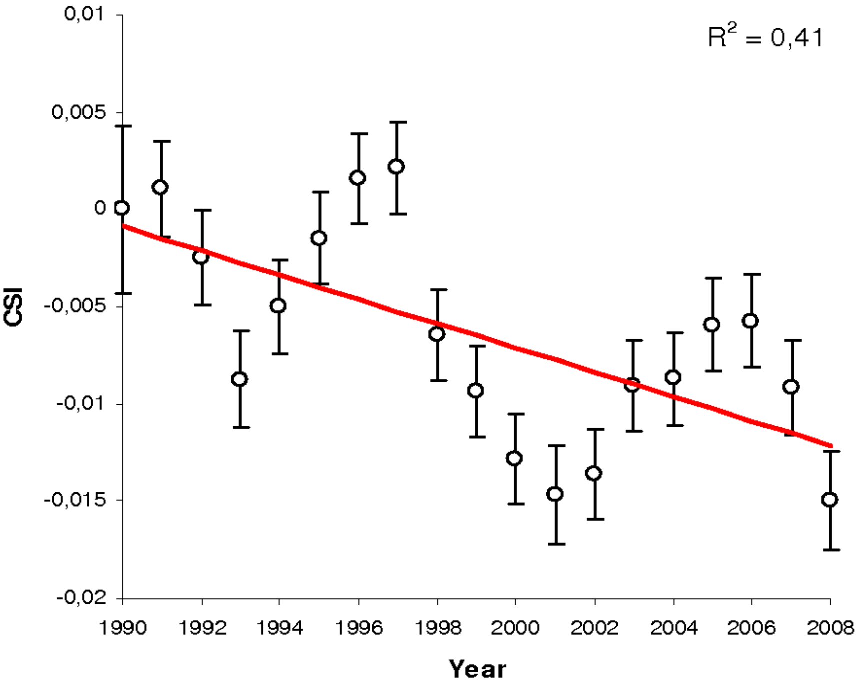
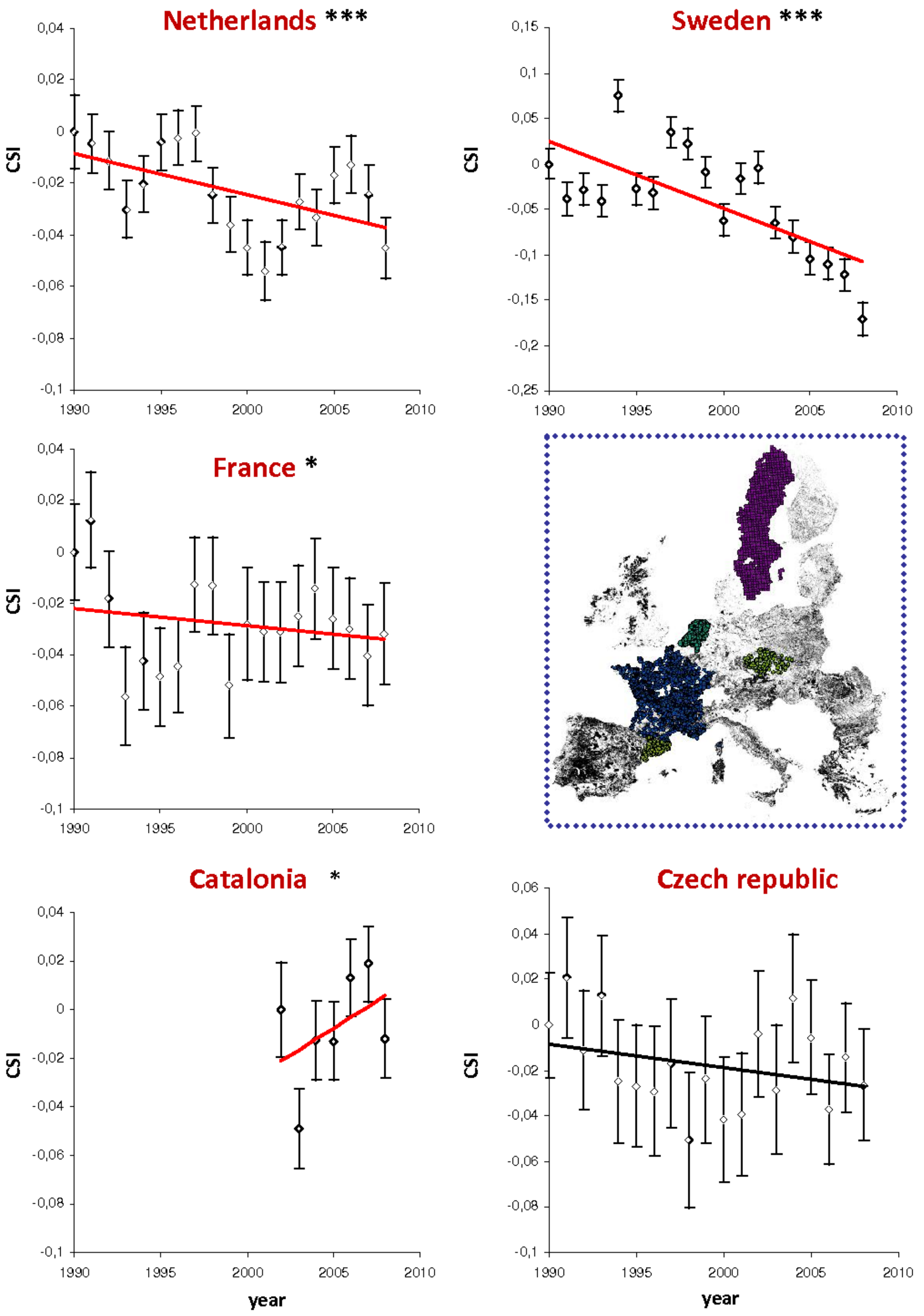


Fig. 11. Significant decline of CSI at European scale through the two last decades as an evidence of the ongoing functional Homogenization process of biodiversity (Project with L. Brotons, A. Lindström, J. Reif, C. Van Turnhout, V. Devictor, F Jiguet, I. Le Viol). Linear mixted models were used, to account in addition to time effect, the effects of site, of country and of sampling effort. CSI ~ factor (Country) + YEAR, random = site , weights = sampling effort). The decline of CSI in Europe was highly significant ($F_{1,25786} = 30,97$, $p < 0.001$).

Interestingly, this decline was consistent across countries (Fig. 12). The CSI decreased significantly in Sweden, France, and Netherlands and showed a negative trend in Czech Republic across the overall period. Some CSI fluctuations seemed to be found simultaneously across some countries (periods of decline alternated with periods of increase). In contrast of other countries, the CSI increased significantly in Catalonia, but note that the surveyed period partly corresponded to a period of a CSI increase in some other countries (Netherlands). Interestingly, this trend in Catalonia was also found in South of France during the same period (pers. Comm. South of France: South of latitude of 46° N).

So our results prove the evidence of an ongoing functional biotic Homogenization process in Europe which is very likely a direct signature of land use change impacts on Biodiversity.



<u>Fig. 12</u>. A consistent trend of CSI across European countries through the two last decades. (Linear mixed models were used to test the significance of trend: Sweden: $F_{1,4503}$=167.58, p <0.001, n = 5239; Republic Czech ns, n= 801; France: $F_{1,5728}$ = 5.4, p= 0.020, n= 7724; Netherlands $F_{1,12669}$ = 17.37, p < 0.001, n = 15002; Catalonia (+) $F_{1,1167}$ = 4.73, p= 0.030, n= 1499 - in project with L. Brotons, A. Lindström, J. Reif, C. Van Turnhout,V. Devictor, F. Jiguet, I. Le Viol).

Trend of CSI across time in Europe is driven by changes in anthropogenic pressures

Many assumptions, both empirical and conceptual, support the view of the relevancy of CSI approach to survey biodiversity states and ecosystem integrity. Of course, the interest of the approach lies in the possibility of having at one's disposal relevant indicator of biodiversity responses facing global changes, especially anthropogenic pressures, to assess the efficiency of environmental policies at European scale. Having shown a significant decline of CSI across Europe across the two last decades using bird standardized collected data, we examined how the decline is explained by land–use pressures changes (Project with: I. Le Viol, V. Devictor, L. Brotons, A. Lindström, J. Reif, C. Van Turnhout, F. Jiguet, A. Doxa, M.L. Parrachini, E. Ivanov - possibly others in the future - work not ever published).

Methods

We used bird CSI per year and per site calculated from the bird database described before (5 countries) and crossed the data with land-use pressures variables, using:

(i) LAND COVER FLOWS databases (LCF1990 and LCF2000)[7]: CLC changes between 1990 and 2000,

(ii) European HNV database[8]: surface area of High nature value farmlands per cell of 1 km² square for Europe in 2000.

Then, in the two cases, we extracted land-use variables for each surveyed site (CSI).

(i) In the case of LCF, we performed a buffer of 2 km radius around each site, clipped the LCF1990 and the LCF2000 databases to get information on type and intensity of CLC changes between the two years per site. The type of changes was characterised using a corresponding table (LCF) provided by ETC-LUSI and intensity of changes was calculated as the overall surface area affected by each type of change per buffer. So we obtained for each site the surface area of each type of CLC changes.

(ii) In the case of HNV, we examined the impact of the proportion of HNV relatively to the area of conventional agriculture on CSI, using two spatial scales: we performed a buffer of 2 km radius and extracted in addition the information per cell of 1 km² that correspond to the location of the site. As the analyses focused on the effects of agriculture “intensity” on CSI trend, we retained the cells which the half of area (50 ha) was occupied by agriculture.

We addressed whether CSI trend per site between 1990 and 2008 was explained by changes in land-use, using linear mixed models. We accounted for the effects of spatial autocorrelation using geographic coordinates as co-variables, effects of country as sampling methods and effort can vary, and accounted for site as random variable.

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[7] Databases provided by ETC-LUSI (Ivanov); http://www.eea.europa.eu/data-and-maps/data/land-cover-flows-based-on-corine-land-cover-changes-database-1990-2000

[8] Databases provided by JRC (Paracchini et al., 2008)

Results

Using buffer of 2 km radius, we did not find significant effects of land use changes, nor across time using LCF dataset, nor across spatial area using HNV database.

In contrast, using information on pressures at smaller scale (1 km²), we find a significant effect of the proportion of HNV. CSI is significantly higher in cell where the proportion of HNV area is higher (Fig. 13). As expected, the bird communities are significantly more specialised where farmland of high nature value are dominant.

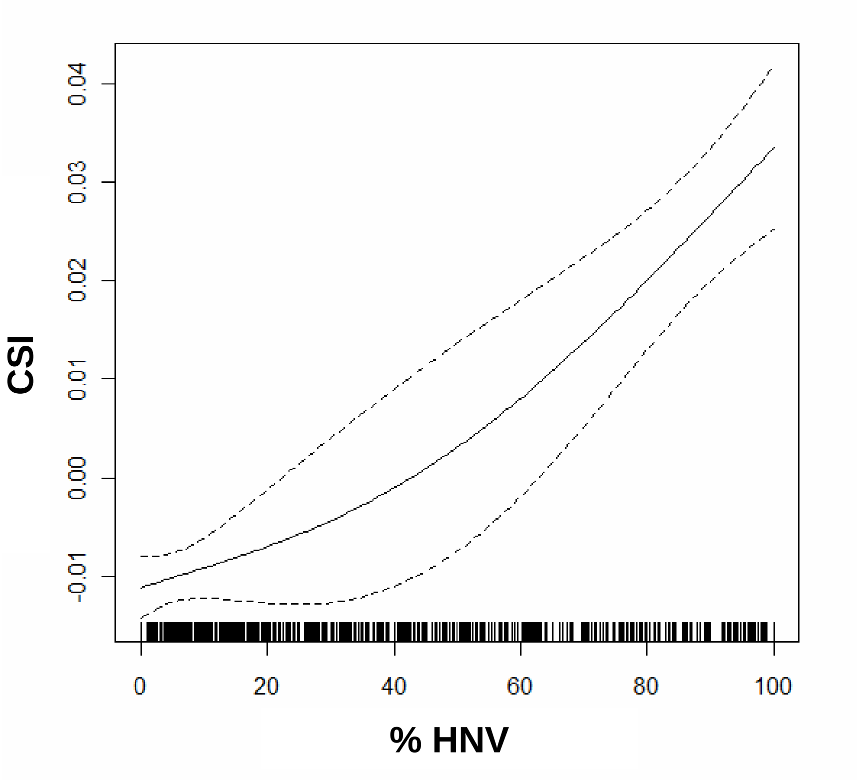

Fig.13. A high positive impact of HNV farmland on the specialisation of bird communities (CSI).

As expected according to the strong global decline of Bird CSI in Europe across the two last decades (p 31), we also found a decline of CSI across time using this dataset. Surprisingly, the CSI tended to decline more in zones dominated by HNV farmland (Hnvperc:YEAR : df =14005 t= -2.07, p= 0.038).

Different explanations can be proposed:

This effect could be due to the initial higher values of CSI in HNV zones. By definition, the CSI of specialized communities (such as it is higher) can decrease stronger than the CSI of generalized communities; therefore the trend can be stronger where CSI is high.

This trend could also be explained by an increase of generalist species due to the withdrawal of agriculture within these zones. For example Kerbiriou et al., (2009) have shown that land-use changes in a multi-protected area (island of Ouessant) have resulted in an increase of the terrestrial breeding bird richness but a decrease of CSI during the last century

(1898-2006). It was due to colonization by generalist species while specialist species have disappeared in the same time.

Note that the method used here is different of those used by Doxa et al (submitted) and Pointereau et al. (2010) who showed that the decline of the CSI for birds has been higher between 1990 and 2008 in France in zones that have lost HNV value between 1970 and 2000. The differences between the results of the two studies could be due to methodology: further analyses are in progress.

Further studies are needed to improve the analyses and better understand the observed trends. Nevertheless, at this time, many evidences show the relevancy of using the CSI approach in addition to richness to survey biodiversity state and ecosystem integrity under global changes at large scale.

Conclusion

- Community Specialisation Index (CSI) is shown to be a very good indicator for biodiversity status and in many cases ecosystem integrity. Ecosystems under stress such as habitat loss, fragmentation or other disturbance including pollution suffer from a decline of specialist species while generalist species could increase, which results in "biotic homogenization process". CSI appears as a good proxy to quantify this biotic homogenization process.
- There is indeed strong conceptual evidence that Community Specialisation, that reflects the relative composition between specialist species with a narrow ecological niche and generalist species which are less requiring, reflects ecosystem stability and thus in large part ecosystem integrity. The approach appears scientifically sound as based on the well-known theoretical concept of the ecological niche.
- Moreover CSI approach presents other advantages: the approach do not imply selection of particular species, nor of particular habitats (all are included) and confirm that observed trend are independent from expert judgment. Interestingly, CSI works with presence-absence data so using large scale inventories which are the most available biodiversity data is feasible to build such biodiversity indicators of ecosystem integrity. And species specialisation expressing the preferences of a species for one or several habitats can be evaluated from different data sources (Expert European database versus standardized collected data from monitoring). From a practical aspect, evaluation of species specialisation based on expertise is less precise but easier than based on monitoring information which is often not available.
- Interestingly, the CSI is linked to habitat degradation mechanistically. It is reactive to changes and much more informative than classic indicators based on species richness or species abundance variations as it encompasses both biodiversity structure and functionalities. It is meaningful for any taxonomic group and meaningful at any spatial and temporal scales.
- Of course, further studies are needed to better understand the CSI trends of different taxonomic groups in Europe at different scales. In order to progress further on this indicator, it is suggested to:
  - explore the variability of species specialisation across their biogeographic range and then develop regional databases of species preferences per habitat to gain in robustness of the indicator ,
  - develop the approach for other taxonomic groups (butterflies, plants), and examine the relevance of the approach for marine and freshwater taxonomic groups.
  - continue to test the sensitivity of the indicator to drivers such as land-use changes in different European countries especially in complement to other indicators (Richness, MSA and other MSTA such as those based on trophic levels).

- Anyway, there are still many evidences which show the relevancy of using the CSI approach to survey biodiversity state and ecosystem integrity under global changes at large scale.

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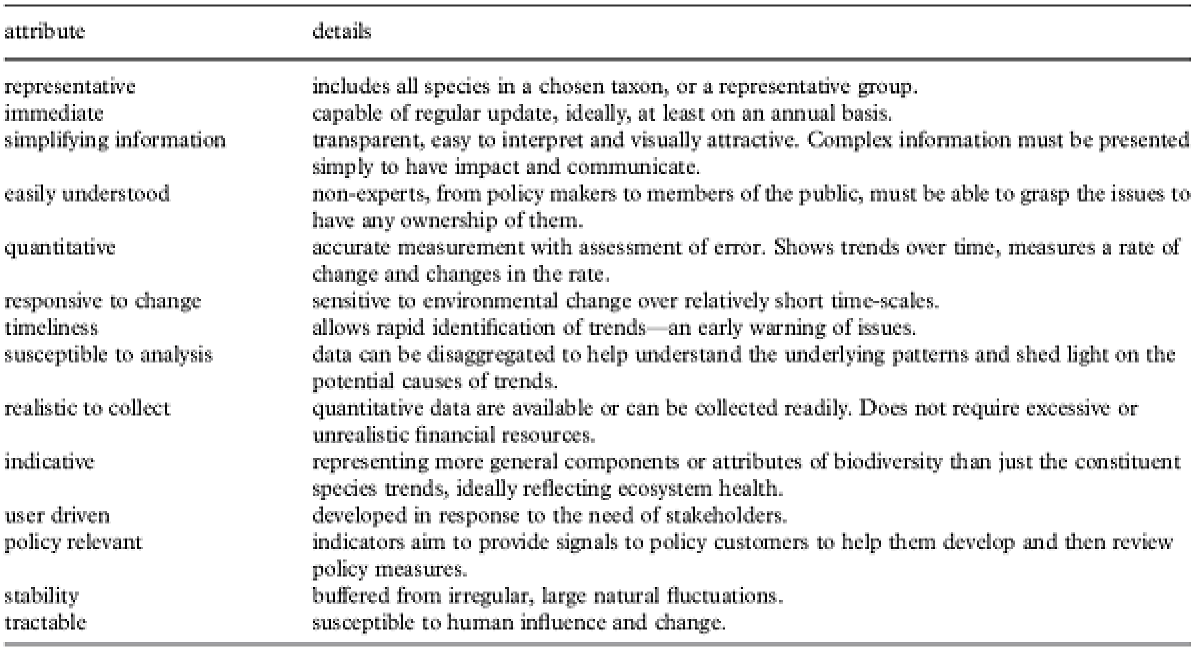
Annexes

Annexe 1. Key attributes of effective indicators of biodiversity

A good biodiversity indicator should have some basic properties. Here is a proposed list of criteria qualifying efficient biodiversity indicators (table bellow from Gregory et al,. 2005).



attribute	details
representative	includes all species in a chosen taxon, or a representative group.
immediate	capable of regular update, ideally, at least on an annual basis.
simplifying information	transparent, easy to interpret and visually attractive. Complex information must be presented simply to have impact and communicate.
easily understood	non-experts, from policy makers to members of the public, must be able to grasp the issues to have any ownership of them.
quantitative	accurate measurement with assessment of error. Shows trends over time, measures a rate of change and changes in the rate.
responsive to change	sensitive to environmental change over relatively short time-scales.
timeliness	allows rapid identification of trends—an early warning of issues.
susceptible to analysis	data can be disaggregated to help understand the underlying patterns and shed light on the potential causes of trends.
realistic to collect	quantitative data are available or can be collected readily. Does not require excessive or unrealistic financial resources.
indicative	representing more general components or attributes of biodiversity than just the constituent species trends, ideally reflecting ecosystem health.
user driven	developed in response to the need of stakeholders.
policy relevant	indicators aim to provide signals to policy customers to help them develop and then review policy measures.
stability	buffered from irregular, large natural fluctuations.
tractable	susceptible to human influence and change.

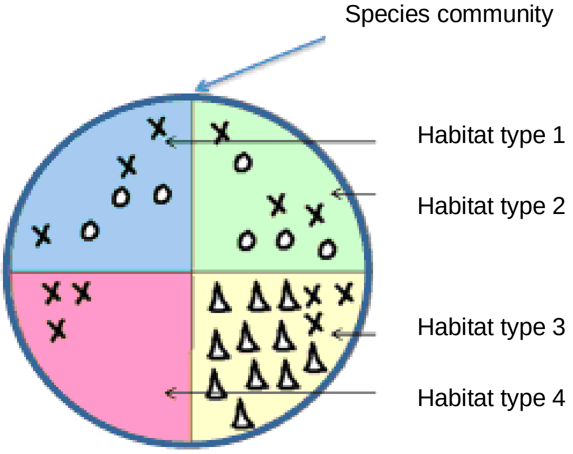
Annexe 2. Summary of calculation methods of SSI and CSI in the case of habitat specialisation

(From Loïs et al., 2009)

Species Specialisation Index		Community Specialisation Index	Resulting information
Data on species' traits preferences		**Data on species within community**	
Estimates from monitoring	+	Densities estimates available	= CSI monitoring/census[1]
		Presence data only	= CSI monitoring/presence
Expert guess work	+	Densities estimates available	= CSI expert/census
		Presence data only	= CSI expert/presence[2]

Community Schema. Consider a species community, in a given area, with three species which show various degrees of specialisation among four habitat types.

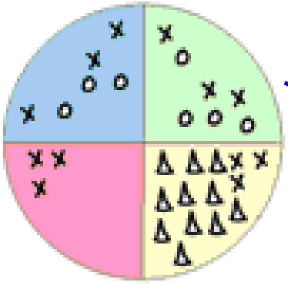
The triangles represent a very specialised species which only uses one habitat type (the yellow one); circles represent less specialised species which use two habitat types (the green and the blue ones); crosses represent very generalist species which use all possible habitats of the studied area (yellow, green, blue and red)



SSI: Species Specialisation Index

SSI estimates from monitoring: Species specialisation index is calculated using quantitative data as the variation coefficient of its abundance across the overall habitat classes.

Mean =μ= (ΣXi)/N Central trend indicator, **Variance** = V2 = (Σ(Xi - μ)²)/N Dispersion indicator, mean of square deviation, **Standard deviation** = σ = √(V2) Mean deviation(same unit as μ), **Variation coefficient** = CV = σ/μ =Dispersion indicator unit less, allowing comparisons between all species, independently from their mean. This Variation coefficient is log-transformed to get a distribution closer to normality

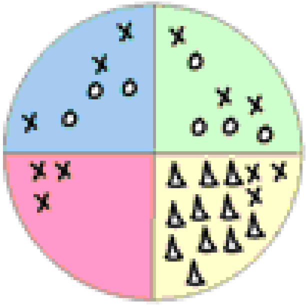


	Habitat type 1	Habitat type 2	Habitat type 3	Habitat type 4	Total
sp ✗	3	3	3	3	12
sp O	3	4	0	0	7
sp Δ	0	0	11	0	11

	Mean	Variance	Standard deviation	Variation coefficient
sp ✗	3	0	0	0
sp O	1.75	3.19	1.79	1.02
sp Δ	2.75	22.69	4.76	1.73

SSI derivates from expertise

H = Total number of habitat types (here, H=4) and **h** = number of habitat types occupied by the species. **h/H** = Proportion of habitat types inhabited by the species i.e. h/H increases linearly with species generalisation. **H/h – 1** = Transformation in order to have a specialisation index rather than a generalisation index thus getting H/h-1 linearly representing species specialisation. **(H/h-1)** ½ = SSI obtained considering it as Coefficient of variation of species densities among habitats (see section bellow) as in this case, habitat with species presence can be considered as habitat of density = 1 and habitats in which species is absent as habitat of species density = 0. Practically, this SSI is log-transformed to obtain a closer distribution to normality.



	Habitat type 1	Habitat type 2	Habitat type 3	Habitat type 4	h
sp ✗	Presence	Presence	Presence	Presence	4
sp O	Presence	Presence	Absence	Absence	2
sp Δ	Absence	Absence	Presence	Absence	1

Species Specialisation Index = $\left(\frac{H}{h}-1\right)^{\frac{1}{2}}$ = SSI

CSI : Community Specialisation Index

CSI is defined as the average specialisation of the species assemblage. The SSI of each species can be weighted by its abundance within the assemblage (CSI census) or not (CSI presence)

$$CSI = \frac{\sum_{i=1}^{n} SSI_i}{n}$$

where n is the number of species within the assemblage.

Annexe 3. Species specialisation Index of birds calculated from Eunis habitat classification.

BIRDS

Species	SSI_Eunis
Accipiter brevipes	9,539
Accipiter gentilis	3,006
Accipiter nisus	2,556
Acrocephalus agricola	5,195
Acrocephalus arundinaceus	4,913
Acrocephalus dumetorum	4,609
Acrocephalus melanopogon	6,480
Acrocephalus paludicola	4,641
Acrocephalus palustris	3,638
Acrocephalus schoenobaenus	3,658
Acrocephalus scirpaceus	6,892
Actitis hypoleucos	3,684
Aegithalos caudatus	2,663
Aegolius funereus	5,797
Aegypius monachus	3,289
Alauda arvensis	3,314
Alca torda	4,814
Alcedo atthis	5,687
Alectoris barbara	3,345
Alectoris chukar	4,759
Alectoris graeca	4,459
Alectoris rufa	6,892
Alle alle	3,469
Anas acuta	2,983
Anas clypeata	3,349
Anas crecca	4,909
Anas penelope	3,523
Anas platyrhynchos	3,591
Anas querquedula	3,997
Anas strepera	4,392
Anser albifrons	3,133
Anser anser	2,574
Anser brachyrhynchus	5,522
Anser erythropus	4,527
Anser fabalis	4,500
Anser rossicus	3,920
Anthropoides virgo	6,289
Anthus berthelotii	3,049
Anthus campestris	2,824
Anthus cervinus	6,480
Anthus hodgsoni	5,401
Anthus petrosus	9,592
Anthus pratensis	2,665
Anthus spinoletta	6,855
Anthus trivialis	2,544
Apus apus	3,893
Apus caffer	6,442
Apus melba	5,622
Apus pallidus	5,537
Apus unicolor	5,772
Aquila adalberti	4,086
Aquila chrysaetos	2,840
Aquila clanga	3,358
Aquila heliaca	3,184
Aquila nipalensis	6,164
Aquila pomarina	2,931
Ardea cinerea	2,435
Ardea purpurea	3,864
Ardeola ralloides	3,192
Arenaria interpres	3,532
Asio flammeus	2,998
Asio otus	2,133
Athene noctua	2,897
Aythya ferina	3,761
Aythya fuligula	4,857
Aythya marila	3,370
Aythya nyroca	4,248
Bombycilla garrulus	8,832
Bonasa bonasia	5,471
Botaurus stellaris	9,798
Branta bernicla	7,874
Branta hrota	7,874
Branta leucopsis	2,879
Bubo bubo	1,979
Bubulcus ibis	3,234
Bucanetes githagineus	2,990
Bucanetes mongolicus	5,415
Bucephala clangula	5,626
Bucephala islandica	4,471
Bulweria bulwerii	5,656
Burhinus oedicnemus	3,196
Buteo lagopus	2,999
Buteo rufinus	4,218
Calandrella brachydactyla	4,170
Calandrella rufescens	4,690
Calcarius lapponicus	9,220
Calidris alba	5,522
Calidris alpina	3,544
Calidris canutus	7,874
Calidris maritima	9,695
Calidris minuta	5,259
Calidris temminckii	4,287
Calonectris diomedea	6,480
Caprimulgus europaeus	2,686

Caprimulgus ruficollis	3,999
Carduelis cannabina	2,587
Carduelis carduelis	3,221
Carduelis chloris	3,221
Carduelis flammea	2,813
Carduelis flavirostris	3,584
Carduelis hornemanni	6,612
Carduelis spinus	9,849
Carpodacus erythrinus	2,443
Carpodacus rubicilla	8,367
Cepphus grylle	3,842
Cercotrichas galactotes	3,337
Certhia brachydactyla	2,917
Certhia familiaris	2,854
Cettia cetti	3,933
Charadrius alexandrinus	2,952
Charadrius asiaticus	3,820
Charadrius dubius	3,656
Charadrius hiaticula	2,981
Charadrius leschenaultii	5,131
Charadrius morinellus	6,819
Chersophilus duponti	4,557
Chettusia gregaria	6,164
Chettusia leucura	6,164
Chlamydotis undulata	6,371
Chlidonias hybridus	5,595
Chlidonias leucopterus	6,782
Chlidonias niger	3,929
Ciconia ciconia	3,064
Ciconia nigra	3,110
Cinclus cinclus	9,849
Circaetus gallicus	2,050
Circus aeruginosus	2,600
Circus cyaneus	3,929
Circus macrourus	3,752
Circus pygargus	2,788
Cisticola juncidis	3,078
Clamator glandarius	3,999
Clangula hyemalis	4,098
Coccothraustes coccothraustes	2,908
Columba bollii	8,062
Columba junoniae	8,062
Columba livia	6,892
Columba oenas	2,052
Columba palumbus	2,626
Columba trocaz	5,656
Coracias garrulus	2,617
Corvus corax	3,979
Corvus corone	2,938
Corvus frugilegus	3,334
Corvus monedula	2,975
Coturnix coturnix	4,220

Crex crex	4,913
Cuculus canorus	1,833
Cuculus saturatus	4,457
Cursorius cursor	5,259
Cyanopica cyana	4,074
Cygnus columbianus	3,839
Cygnus cygnus	3,073
Cygnus olor	2,691
Delichon urbica	4,358
Dendrocopos leucotos	7,031
Dendrocopos major	2,940
Dendrocopos medius	4,363
Dendrocopos minor	4,382
Dendrocopos syriacus	2,753
Dryocopus martius	5,114
Egretta alba	3,864
Egretta garzetta	2,900
Elanus caeruleus	5,227
Emberiza aureola	6,480
Emberiza bruniceps	6,164
Emberiza buchanani	4,829
Emberiza caesia	3,337
Emberiza cia	2,989
Emberiza cineracea	5,227
Emberiza cirlus	2,807
Emberiza citrinella	3,300
Emberiza hortulana	2,024
Emberiza leucocephalos	5,291
Emberiza melanocephala	2,954
Emberiza pallasi	7,874
Emberiza pusilla	6,324
Emberiza rustica	5,032
Emberiza schoeniclus	4,582
Eremophila alpestris	4,239
Erithacus rubecula	2,477
Falco biarmicus	5,227
Falco cherrug	3,073
Falco columbarius	3,886
Falco eleonorae	5,259
Falco naumanni	4,215
Falco peregrinus	2,364
Falco rusticolus	5,259
Falco subbuteo	2,736
Falco tinnunculus	2,418
Falco vespertinus	3,524
Ficedula albicollis	3,595
Ficedula hypoleuca	4,358
Ficedula parva	5,326
Ficedula semitorquata	3,911
Francolinus francolinus	4,499
Fratercula arctica	4,787
Fringilla coelebs	2,770

Fringilla montifringilla	4,769
Fringilla teydea	8,062
Fulica atra	5,151
Fulica cristata	4,809
Fulmarus glacialis	6,708
Galerida cristata	2,717
Galerida theklae	3,661
Gallinago gallinago	5,627
Gallinago media	4,663
Gallinago stenura	5,259
Gallinula chloropus	4,909
Garrulus glandarius	3,550
Gavia adamsii	2,964
Gavia arctica	2,744
Gavia immer	4,471
Gavia stellata	2,896
Gelochelidon nilotica	2,908
Glareola nordmanni	3,861
Glareola pratincola	3,711
Glaucidium passerinum	4,174
Grus grus	2,793
Gypaetus barbatus	5,415
Gyps fulvus	5,415
Haematopus ostralegus	2,577
Haliaeetus albicilla	2,689
Hieraaetus fasciatus	3,514
Hieraaetus pennatus	4,743
Himantopus himantopus	3,351
Hippolais caligata	3,073
Hippolais icterina	3,226
Hippolais languida	3,786
Hippolais olivetorum	6,442
Hippolais pallida	3,151
Hippolais polyglotta	3,172
Hirundo daurica	4,499
Hirundo rustica	3,286
Histrionicus histrionicus	5,522
Hoplopterus spinosus	2,768
Hydrobates pelagicus	6,964
Iobrychus minutus	6,994
Irania gutturalis	5,227
Jynx torquilla	3,903
Lagopus lagopus	3,573
Lagopus mutus	4,769
Lanius collurio	3,414
Lanius excubitor	3,870
Lanius meridionalis	2,984
Lanius minor	3,343
Lanius nubicus	3,535
Lanius senator	2,584
Larus argentatus	2,501
Larus armenicus	6,164

Larus audouinii	5,311
Larus cachinnans	3,457
Larus canus	2,714
Larus fuscus	3,258
Larus genei	3,337
Larus glaucoides	3,404
Larus hyperboreus	2,826
Larus ichthyaetus	3,463
Larus marinus	3,106
Larus melanocephalus	3,486
Larus michahellis	3,018
Larus minutus	5,567
Larus ridibundus	2,345
Larus sabini	5,522
Limicola falcinellus	9,381
Limosa lapponica	9,381
Limosa limosa	4,804
Locustella fluviatilis	4,166
Locustella lanceolata	5,873
Locustella luscinioides	6,892
Locustella naevia	3,907
Loxia curvirostra	7,031
Loxia leucoptera	8,832
Loxia pytyopsittacus	9,487
Loxia scotica	9,110
Lullula arborea	2,540
Luscinia calliope	8,367
Luscinia luscinia	2,959
Luscinia megarhynchos	2,575
Luscinia svecica	3,377
Lymnocryptes minimus	5,032
Marmaronetta angustirostris	3,705
Melanitta fusca	3,042
Melanitta nigra	3,110
Melanocorypha bimaculata	4,170
Melanocorypha calandra	4,499
Melanocorypha leucoptera	8,775
Mergus albellus	4,329
Mergus merganser	4,287
Mergus serrator	3,368
Merops apiaster	3,315
Merops superciliosus	3,820
Miliraria calandra	3,936
Milvus migrans	2,574
Milvus milvus	2,660
Monticola saxatilis	3,617
Monticola solitarius	4,358
Montifringilla nivalis	4,311
Morus bassanus	6,708
Motacilla alba	2,487
Motacilla cinerea	9,849
Motacilla citreola	5,537

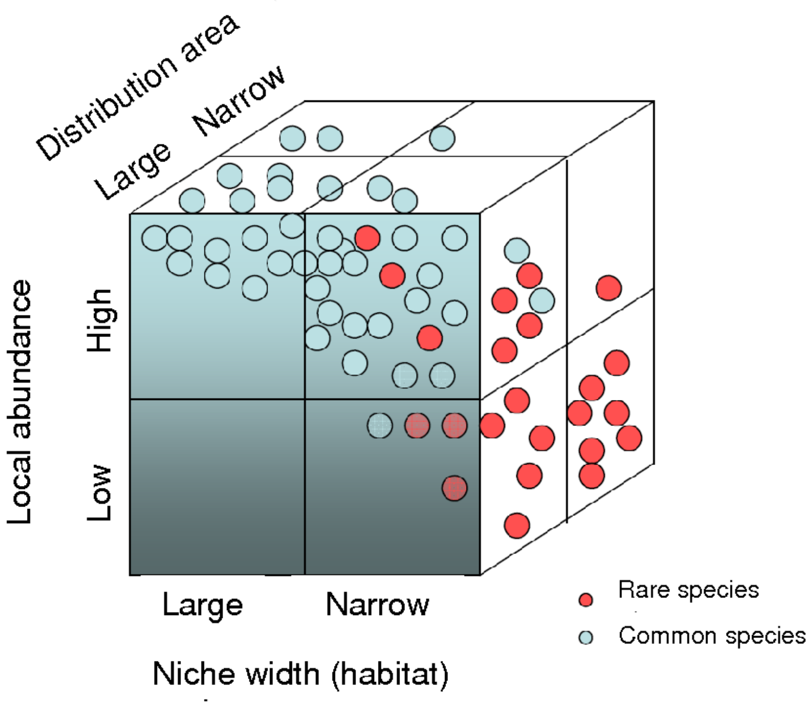
Motacilla flava	3,704
Muscicapa striata	2,303
Neophron percnopterus	4,242
Netta rufina	5,507
Nucifraga caryocatactes	6,881
Numenius arquata	3,395
Numenius phaeopus	3,713
Nyctea scandiaca	6,480
Nycticorax nycticorax	3,892
Oceanodroma castro	6,633
Oceanodroma leucorhoa	4,170
Oenanthe cypriaca	5,227
Oenanthe finschii	6,324
Oenanthe isabellina	5,507
Oenanthe leucura	5,502
Oenanthe melanoleuca	2,631
Oenanthe oenanthe	2,487
Oenanthe pleschanka	4,223
Oenanthe xanthoprymna	5,958
Oriolus oriolus	2,514
Otis tarda	5,446
Otus scops	2,529
Oxyura leucocephala	4,074
Pagophila eburnea	2,826
Pandion haliaetus	3,951
Panurus biarmicus	6,892
Paru caeruleus	2,659
Parus ater	3,604
Parus cinctus	6,519
Parus cristatus	6,892
Parus cyanus	5,259
Parus lugubris	3,072
Parus major	2,436
Parus montanus	3,143
Parus palustris	3,720
Passer domesticus	2,302
Passer hispaniolensis	3,090
Passer moabiticus	3,686
Passer montanus	3,338
Pelagodroma marina	5,656
Pelecanus crispus	5,446
Pelecanus onocrotalus	4,499
Perdix perdix	3,571
Perisoreus infaustus	9,274
Pernis apivorus	5,597
Petronia brachydactyla	6,442
Petronia petronia	4,368
Phalacrocorax aristotelis	6,928
Phalacrocorax carbo	3,633
Phalacrocorax pygmeus	4,215
Phalaropus fulicarius	5,522
Phalaropus lobatus	5,537

Philomachus pugnax	4,830
Phoenicopterus ruber	3,688
Phoenicurus ochruros	3,395
Phoenicurus phoenicurus	2,385
Phylloscopus bonelli	3,082
Phylloscopus borealis	4,649
Phylloscopus collybita	2,892
Phylloscopus inornatus	6,595
Phylloscopus lorenzii	9,327
Phylloscopus nitidus	6,557
Phylloscopus sibilatrix	2,905
Phylloscopus trochiloides	4,716
Phylloscopus trochilus	3,740
Pica pica	2,611
Picoides tridactylus	6,729
Picus canus	4,339
Picus viridis	2,364
Pinicola enucleator	6,330
Platalea leucorodia	3,864
Plectrophenax nivalis	3,544
Plegadis falcinellus	4,112
Pluvialis apricaria	4,242
Pluvialis squatarola	7,874
Podiceps auritus	5,413
Podiceps cristatus	4,382
Podiceps grisegena	6,892
Podiceps nigricollis	6,892
Porphyrio porphyrio	4,593
Porzana parva	9,695
Porzana porzana	9,798
Porzana pusilla	5,595
Prunella atrogularis	8,367
Prunella collaris	5,567
Prunella modularis	2,069
Prunella montanella	3,973
Prunella ocularis	9,000
Pterocles alchata	5,311
Pterocles orientalis	4,621
Pterodroma feae	5,656
Pterodroma madeira	5,656
Ptyonoprogne rupestris	5,622
Puffinus assimilis	3,936
Puffinus mauretanicus	6,442
Puffinus puffinus	6,708
Puffinus yelkouan	6,442
Pycnonotus xanthopygos	5,227
Pyrrhocorax graculus	4,804
Pyrrhocorax pyrrhocorax	3,584
Pyrrhula murina	8,062
Pyrrhula pyrrhula	3,684
Rallus aquaticus	7,031
Recurvirostra avosetta	3,013

Regulus ignicapillus	3,263
Regulus regulus	4,795
Remiz pendulinus	6,892
Rhodopechys sanguinea	3,686
Riparia riparia	3,676
Rissa tridactyla	6,708
Saxicola dacotiae	3,558
Saxicola rubetra	3,700
Saxicola torquata	2,824
Scolopax rusticola	4,381
Serinus canaria	2,207
Serinus citrinella	5,507
Serinus pusillus	3,532
Serinus serinus	2,419
Sitta europaea	3,220
Sitta krueperi	6,805
Sitta neumayer	6,670
Sitta tephronota	5,958
Sitta whiteheadi	9,165
Somateria mollissima	3,146
Somateria spectabilis	2,619
Stercorarius longicaudus	9,220
Stercorarius parasiticus	6,745
Stercorarius pomarinus	5,522
Stercorarius skua	5,446
Sterna albifrons	3,379
Sterna caspia	3,807
Sterna dougallii	4,199
Sterna hirundo	2,592
Sterna paradisaea	3,296
Sterna sandvicensis	3,638
Streptopelia decaocto	2,686
Streptopelia turtur	2,473
Strix aluco	2,722
Strix nebulosa	4,329
Strix uralensis	5,384
Sturnus roseus	3,239
Sturnus unicolor	3,055
Sturnus vulgaris	3,100
Surnia ulula	4,668
Sylvia atricapilla	2,641
Sylvia borin	2,931
Sylvia cantillans	5,533
Sylvia communis	3,999
Sylvia conspicillata	4,024
Sylvia curruca	2,472
Sylvia hortensis	3,456
Sylvia melanocephala	2,871
Sylvia melanothorax	9,165
Sylvia mystacea	4,677
Sylvia nisoria	2,685
Sylvia rueppelli	4,593
Sylvia sarda	6,572
Sylvia undata	4,344
Tachybaptus ruficollis	6,892
Tadorna ferruginea	3,561
Tadorna tadorna	3,633
Tarsiger cyanurus	9,274
Tetrao mlokosiewiczi	5,065
Tetrao tetrix	3,544
Tetrao urogallus	5,653
Tetraogallus caspius	4,760
Tetraogallus caucasicus	4,760
Tetrax tetrax	5,595
Tichodroma muraria	6,994
Tringa erythropus	3,799
Tringa glareola	4,867
Tringa nebularia	3,544
Tringa ochropus	4,289
Tringa stagnatilis	4,191
Tringa totanus	2,914
Troglodytes troglodytes	2,155
Turdus iliacus	3,291
Turdus merula	2,234
Turdus philomelos	2,511
Turdus pilaris	4,847
Turdus ruficollis	5,993
Turdus torquatus	3,182
Turdus viscivorus	3,143
Turnix sylvatica	5,227
Tyto alba	2,665
Upupa epops	2,555
Uria aalge	4,814
Uria lomvia	3,920
Vanellus vanellus	4,847
Xenus cinereus	3,292
Zoothera dauma	8,367

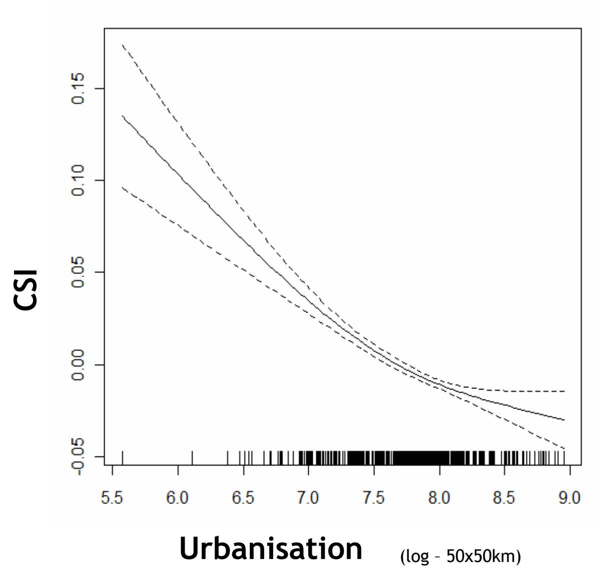
Annexe 4. Definition criteria of rare and common species

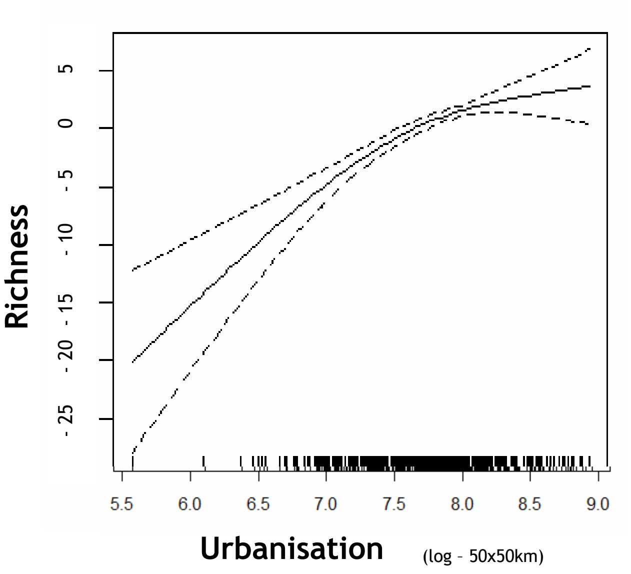
(modified from Rabinowitz 1981)

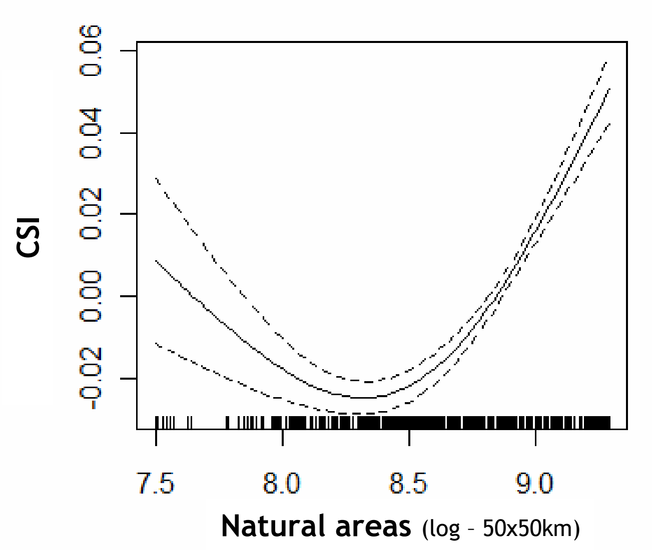


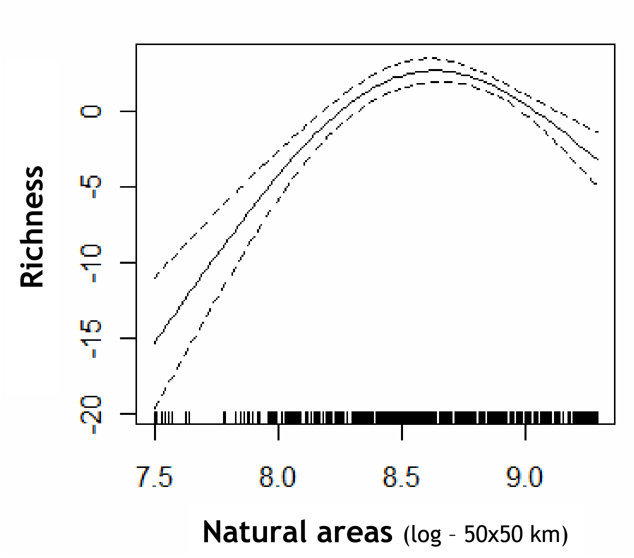
Annexe 5. Responses of mammals CSI and richness to land use pressures intensity in Europe (altitude > 150 meters).

Analyses were performed using generalized least squares models (see text). Urbanization corresponds to the overall area of urban area per cell, and natural area is the overall natural area per cell. These variables were log transformed. Urbanisation: CSI: negative trend, p=0.06, Richness: positive trend, p<0.001. Natural areas: CSI: quadratic effect with positive trend, p = 0.04, Richness quadratic effect, positive trend p<0.001









Annexe 6. Bird Communities Specialisation Index (n = 483 species)

The higher is the CSI, the more specialized is the community