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Annex II: Review of Resilience Indicators

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Contents

1 REVIEW OF POTENTIAL ECOSYSTEM RESILIENCE INDICATORS 4

1.1 A. Resilience is positively related to species richness 4

1.1.1 Species level 4

1.1.2 Community level 6

1.2 B. Resilience increases with complexity 7

1.3 C. Resilience increases with (meta) population size 8

1.4 D. Resilience increases with habitat area 9

1.5 E. Resilience increases with ecosystem connectivity 10

1.6 F. Resilience increases with ecosystem condition 14

1 REVIEW OF POTENTIAL ECOSYSTEM RESILIENCE INDICATORS

This Annex provides a detailed review of existing indicators, in particular the SEBI set, with respect to their relevance to ecosystem resilience, as discussed above, and their practicality, taking into account the necessary properties of efficient indicators (as summarised in Box 4.5). The assessment also took into account the findings of the Green Infrastructure Expert Workshop[1] held by the Commission on the 7 September 2011 (see Box 4.6).

The Indicators are grouped according to the ecosystem property and resilience assumptions described in chapter 4, and further details of the review and a summary of these findings are provided in section 4.2.2.

1.1 A. Resilience is positively related to species richness

Under this assumption existing indicators can be used to measure the degree of resilience according to two levels of organisation:

1.1.1 Species level

A. The most simple and straightforward indicator is *Species richness*. The basic idea behind this indicator is that species richness is positively related to redundancy and insurance: if one particular species fails, its function is taken over by another, more or less equivalent one (Gunderson, 2000). In practice, data are only available for a limited number of taxonomic groups.

   SEBI indicators that can potentially be used: *Abundance and distribution of selected species*[2]. This indicator provides a time-series aggregated index of population change, but currently only covers common farmland and forest birds and grassland butterflies and therefore is far from a representative sample of taxonomic groups. This may generate significant bias because trends may differ in different groups. Therefore, there is a potential risk that the indicators may not reflect changes in other groups, many of which might be of greater importance in terms of increase ecosystems resilience (eg plants and invertebrates). Furthermore, the indicator does not directly measure species richness, although it is likely to be correlated to some degree to the indexes. We therefore assess the applicability of this indicator to assess ecosystem resilience as *low-moderate.*

B. A more complicated indicator is the *occurrence of rare species* as listed in Red Lists. Data availability of rare species belonging to the above-mentioned taxonomic groups is likely to be at least as good as for all species (and probably better) but the relationship with resilience is less clear. In many habitats the number of rare and presumably sensitive, species may well be an indicator of habitat quality but there is also a (probably

[1] Ecologic, GHK & IEEP (2011) Green Infrastructure Expert Workshop, (7/9/2011) Summary of Working Groups

[2] http://www.eea.europa.eu/data-and-maps/indicators/abundance-and-distribution-of-selected-species/abundance-and-distribution-of-selected

considerable) group of rare organisms that are restricted to ephemeral habitats. Such habitats are by definition unstable and not resilient.

SEBI indicators that can potentially be used: *Red list Index for European Species* and *Species of European Interest*[3]. The indicator provides an aggregated measure of the change in status of taxa between two or more Red List assessments. Again it does not provide a direct measure of species richness, though declines in in the index will probably indicate declines in species richness to some degree. It currently only covers birds, so suffers from the limitations mentioned above. However, regional Red List assessments are increasingly being carried out for other taxa (eg amphibians and reptiles) and therefore the index could be calculated for other species when repeat assessments are carried out in future.

We assess the applicability of this indicator to assess ecosystem resilience as *low.*

C. Intuitively the presence of *keystone species* seems a good indicator of ecosystem robustness and resilience. Keystone species have a disproportionate impact on their ecosystem and are therefore closely related to its functioning. Removal of such species can lead to a cascade of unwanted effects and often to a catastrophic shift (Mclaren, 1994; Mills, 1993; Paine, 1969). Unfortunately, the term is "broadly applied and poorly defined" (Mills, 1993) and for most habitats it is unknown which species have a disproportionate impact or whether the role of such species can be taken over by other ones if they disappear.

SEBI indicators that can potentially be used: none

D. Data on the *stability of species' population and/or meta-population size* (eg of keystone species or Red List species) or, conversely, species turnover rates, seem an excellent species-level indicator of ecosystem resilience because this parameter quantifies actual changes. In order to be meaningful such an indicator would require data at the NUTS 2 regional level or at higher resolutions because trends can vary considerably geographically. For example, long-term trends in the abundance of a few dozen critical bird species have been observed to differ significantly between different regions even within a small country like the Netherlands (Van Kleunen et al. 2005). Moreover, care should be taken that species used in the indicator are taken from different systematic groups because trends may differ completely between taxonomic groups. In reality, however, such data are even in well-investigated regions only available for a few taxonomic groups, such as birds and even then often only for a limited number of habitats (farmland, wetlands, forests). Typically such data are comparatively easily available in a low spatial resolution whereas higher spatial resolution data have to be acquired from a large range of sources. Developments in data storage costs and on-line access possibilities are, however, rapidly changing and it is likely that in several countries large databases will be accessible from the desktop within five to 10 years.

[3] http://www.eea.europa.eu/data-and-maps/indicators/red-list-index-for-european-species/red-list-index-for-european

SEBI indicators that can potentially be used: *Abundance and distribution of selected species*. The same risks exist as mentioned under A. We assess the practical applicability of this indicator to assess ecosystem resilience as *low-moderate.*

1.1.2 Community level

E. An alternative to using species richness as a resilience indicator is to calculate a so-called *Saturation index* (Wolters et al, 2005) to measure the "intactness" of the community. The underlying assumption is that more intact communities are more resilient than impoverished ones. Theoretically this indicator is a better reflection of ecosystem "completeness" than mere species richness because even under undisturbed conditions there are large differences in species richness between different communities. This does not necessarily imply that the communities poorer in species are more degraded and less resilient. A critical step in the application of this index is to define precisely the community and its constituent species. In principle, such an approach is even more data intensive than indices based on species richness, but it can be simplified by only calculating the index for the habitats of Community importance listed in Annex 1 of the Habitats Directive together with the characteristic species mentioned therein. Whether such a simplified approach is sensitive enough to detect relevant differences between sites remains to be investigated with the help of sample data sets, but studies analysing the effectiveness of alternative restoration techniques (e.g. Klimkowska et al, 2007) have applied this indicator successfully.

   SEBI indicators that can potentially be used: None, but data gathered to calculate the indicator *Species of European interest*[4] can potentially be classified to the habitat types they belong to. If there are enough species per habitat type this should enable an assessment of the above-mentioned saturation index. However, this will be the case for only a very limited number of habitat types and even then, only a small fraction of the species will be covered. This implies that the reliability of this potentially powerful indicator will be very limited. We therefore assess the current applicability of this indicator to assess ecosystem resilience as *poor.* Future prospects of this indicator may be much better if increased monitoring of species occurs because of fast developments in centralising data storage and accessibility in many European countries.

F. A somewhat similar approach is to use *indices related to the degree of deviation with the undisturbed situation* such as the *Natural Capital index* (NCI) (Brink B.J.E.ten, 2002) and the *Mean Species Abundance* (MSA) (Alkemade et al, 2009). NCI assesses the difference in between natural conditions and the actual situation in terms of species composition, species abundance and quality and extent of the area. Alternatively, MSA and its derivatives are not based on actual data but instead use known relationships between pressures and impacts on species abundance to predict MSA levels, which relate to estimated changes in biodiversity from that expected to occur in the original natural system. Again the assumption with respect to resilience is that there is a positive relation

[4] http://www.eea.europa.eu/data-and-maps/indicators/species-of-european-interest/species-of-european-interest-assessment

between the naturalness of a system and the amount of feedbacks it contains, ie its resilience. The NCI has only been applied in exploratory studies (Brink B.J.E.ten, 2002) and is nowadays exclusively used as a descriptive, non-quantitative term. The MSA is now commonly used in global level in studies by UNEP, CBD and OECD (Bakkes et al, 2011), but its applicability to Europe is questionable due to its reference to changes from natural systems, whereas Europe is now dominated by man-made and semi-natural habitats. Furthermore, the predictions are based on rather few broad pressures, such that many of the more subtle impacts of land use change that occur in Europe, but which nevertheless have significant impacts on biodiversity, are not considered.

SEBI indicators that can potentially be used: None.

1.2 B. Resilience increases with complexity

G. Direct measurements of functional complexity-related parameters are not generally available for most taxonomic groups but in the case of vascular plants it is possible to estimate functional richness on the basis of a combination of species lists (eg in a national grid) and databases of plant traits (Kattge, 2011; Kleyer et al, 2008; Villeger, 2008). Because of the positive relationship between species richness and functional diversity (Petchey and Gaston, 2002), a simpler approach is to use species-richness of selected groups per (NUTS 2) region and habitat type as a proxy. An alternative proxy is to calculate a "saturation index" (see *assumption 1 heading* E) for one taxonomic group (plants, butterfies) per habitat type and region because available evidence suggests that functional complexity increases with the "completeness" of the ecosystem (Laliberte et al, 2010). Species lists should be partly available from data gathered in relation to monitoring obligations on the conservation status of species and natural habitats under the Habitats Directive. But data may also need to be sought from local inventories etc.

SEBI indicators that can potentially be used: None.

Closely related to the above-mentioned relationship between functional diversity and ecosystem resilience is the assumption that resilience increases with *stability, complexity and length of food webs*. Unfortunately, there is much less research available on this topic and the few publications that are available do not support this assumption (Vallina and Le Quere, 2011). Moreover, although there are many studies on, often simplified, food webs there is not much knowledge on the structure of actual food webs (Girvan and Newman, 2002), nor are they easily measurable and many are likely to be dynamic in time and space. We therefore do not suggest searching for an indicator of food web complexity as a proxy for ecosystem resilience.

SEBI indicators that can potentially be used: None.

H. Most ecologists assume that there is a close relationship between *structural* diversity and *functional* diversity. Hence, there should also be a close relationship between structural diversity and ecosystem resilience. Since there is also a close correlation between structural diversity and spatial heterogeneity, the latter can potentially be used as an indicator of ecosystem resilience, especially as heterogeneity can be relatively easily

measured in spatial images. Indeed, theoretical studies find such a relationship (Van Nes and Scheffer, 2005) although practical evidence is scarce (Virah-Sawmy et al, 2009). This is potentially a very suitable and powerful indicator, especially as data availability is very good. Easily accessible aerial photographs in several wavelengths are available for large surfaces, both of the current situation and of up to some decades ago meaning that analyses of developments over time can be made. GIS applications provide several potentially suitable indices.

SEBI indicators that can potentially be used: None.

Although no SEBI or other indicators of ecosystem complexity are currently known to be in use their development does seem plausible. The basic data that are necessary to calculate functional richness and spatial heterogeneity are sufficiently available for larger surfaces. The quality of the spatial data does not differ much between countries but this does differ for data related to functional complexity. Nevertheless, it is even possible to assess this parameter on the basis of simple data such as species lists from descriptive reports, national distribution maps or similar data sources, provided these are relatively reliable. Of course, the spatial resolution of the calculated parameters can never be higher than the resolution of the original data they are based upon. In the light of all this, both indices are therefore potentially good candidates to assess (changes in) ecosystem resilience for defined areas. The drawback of both parameters is that they need much more pre-processing of data than much simpler indicators based upon biodiversity-related parameters. They also need more expert-input to evaluate their meaning.

1.3 C. Resilience increases with (meta) population size

The most critical point in this indicator is to determine the size of the actual meta-population and the gene-flow between sub-populations. Nevertheless, this is a potentially promising indicator, provided the results are interpreted carefully. Some of the problems mentioned could be partially overcome by using proxies such as density per unit area and trends therein (animals) or frequency of occurrence in a grid system (plants) instead of actual population sizes. As for many other indicators mentioned, there are only relevant data for a few selected taxonomic groups: birds (censuses etc. are available at least at the level of province), higher plants (presence/absence data in a grid system, for many countries available for at least two time periods), and, in several countries, data on certain insect groups (butterflies, dragon flies, carabid beetles) are available as well.

SEBI indicators that can be potentially used: *Abundance and distribution of selected species*. The basic data used to calculate this index can be used to determine the above-mentioned proxies and in that way give an indication of (changes in) population density of at least a limited number of taxonomic groups. Because of the limited number of groups covered, we assess the applicability of this indicator to assess ecosystem resilience as *low-moderate.*

This indicator provides a time-series aggregated index of population change, but currently only covers common farmland and forest birds and grassland butterflies and therefore is far from a representative sample of taxonomic groups. This may generate significant bias because trends may differ in different groups. Therefore, there is a potential risk that the

indicators may not reflect changes in other groups, many of which might be of greater importance in terms of increased ecosystem resilience (eg plants and invertebrates). Furthermore, the indicator does not directly measure species richness, although it is likely to be correlated to some degree to the indexes. We therefore assess the applicability of this indicator to assess ecosystem resilience at present as low-moderate but future prospects seem much better. For a further discussion: see 1-B

1.4 D. Resilience increases with habitat area

Unfortunately there is no easy answer to the question *how large an area of habitat is required to ensure a reasonably resilient ecosystem?* Resilience is not driven by the identity or size of any given element of the system, but rather by the functions those elements provide, and their distribution within and across scales (Allen et al, 2005). However, the number of functions and thus resilience, is very likely to increase with area. Equivalent to indicators used to assess trends in global biodiversity (Butchart et al, 2010), we propose to compare areas for this indicator on the basis of simple metrics such as the extent of forest, wetlands, high-natural value farmland and similar, and the pattern of such land classes (spatial configuration into patches of different sizes) in the landscape. The great advantage of this indicator is that categories can be defined according to the needs of the user and the quality and availability of the data. Relatively broadly defined classes seem sufficient to suit policy needs at the European level. Data can be retrieved from databases like CORINE, EUNIS and national databases.

<u>SEBI indicators that can be potentially used</u>:
Fragmentation of natural and semi-natural areas[5]. This indicator consists of 3 sub-indicators: a landscape mosaic sub-indicator on the degree of fragmentation of natural and semi-natural areas, a core forest fragmentation sub-indicator looking at the splitting apart of core forest patches, and a forest connectivity sub-indicator. In the present case the landscape mosaic sub-indicator (Estreguil and Caudullo 2011) is sufficient. The indicator aims to answer where, how and how much, a piece of natural land is surrounded by other "natural" lands, or is intermingled with agricultural and artificial lands. The index reports per region (landscape unit, province or country) the share of natural land in a fragmented (and unfragmented) pattern type, as well as the average patch size per patch size ranges (below 10 hectares, in between 10-100 hectares, above 100 hectares) of "unfragmented" natural/semi-natural patches. This indicator enables to report trends on the basis of input land cover mapsof several points in time. See eg Estreguil and Caudullo 2011.

From the qualitative thematic point of view, input CLC data have some limitations with respect to defining degree of naturalness of the classes (for example, the CLC forest class includes introduced species, mono-species plantations, etc) and the scale of observation (heterogeneity of some classes such as agriculture with natural vegetation). In situ data would be needed to refine this. We assess the applicability of this indicator to assess

[5] http://www.eea.europa.eu/data-and-maps/indicators/fragmentation-of-natural-and-semi/fragmentation-of-natural-and-semi ; This indicator itself was developed by the Joint Research Center (JRC) (update on indicator and derived products are at http://forest.ec.europa.eu/forest-pattern and http://efdac.jrc.ec.europa.eu/pattern/map/

ecosystem resilience theoretically as *fair-good* but are aware of limitations in the underlying input data. The model used for this indicator is now automated in GIS environment at JRC and can easily be run with any input data.

1.5 E. Resilience increases with ecosystem connectivity

As discussed above, our understanding of the relationship between landscape connectivity and ecosystem resilience is still in its infancy. Moreover, there is debate over which *structures* are most important for *connectivity* (eg Goodwin and Fahrig, 2002b). Consequently, there is also no agreement on which metrics to use to evaluate landscape connectivity, although there is growing consensus that the measurement of *functional* connectivity is of particular importance, and that this differs from species to species. A given landscape may have different degrees of connectivity for different species (Kindlmann and Burel, 2008). This is of course highly impractical for policy actions aimed at conserving or restoring certain landscape *structures*. Beier et al (2011) approached this problem from a policy viewpoint and analysed what type of connectivity metrics are useful for practical decisions. In fact they went back to structural connectivity and suggested using simple metrics for large-scale conservation planning rather than complex ones. Their study stresses the need to explicitly define which goal(s) increasing coherence should have because these define the required scale and type of connectivity (contiguous, stepping stones) to be conserved and/or restored.

For practical reasons the present discussion focusses on connectivity for large(r) organisms. Although many plant species may be impacted by fragmentation (Hermy, 1999) so far most published studies on fragmentation have focused on large mammals and birds. This can be justified by the assumption that large organisms themselves are indicative of resilient ecosystems, or are at least a good transport vector for other organisms such as the seeds of vascular plants (Mouissie, 2005), and thus for a functioning metapopulation of the latter organisms as well.

For the sake of clarity, most indicators like the ones in SEBI *Fragmentation of natural and semi-natural areas*[6] provide connectivity related metric applied to a single ecosystem, eg changes in forest connectivity or the fragmentation of river systems. Unfortunately, this does not necessarily mean that they describe a general trend accounting for all ecosystems, particularly because changes in connectivity may be partly interchangeable between habitats. For instance, in Northwest Europe there was a significant increase in forest connectivity over the last century at the cost of an even larger decrease in heathland (Ellenberg 1988; Webb, 1990; Bakker & Van Diggelen 2006; Härdtle et al. 2009). Forest-based fragmentation and connectivity indicators would show a positive trend whereas the actual development should be evaluated negatively because a priority habitat ("heathland") is replaced by a non-priority habitat ("pine plantation"). In addition to ecosystem specific

[6] http://www.eea.europa.eu/data-and-maps/indicators/fragmentation-of-natural-and-semi/fragmentation-of-natural-and-semi. ; This indicator itself was developed by the Joint Research Center (JRC) (update on indicator and derived products are at http://forest.ec.europa.eu/forest-pattern and http://efdac.jrc.ec.europa.eu/pattern/map/). It consists of 3 sub-indicators (landscape mosaic, core forest fragmentation, forest connectivity)

connectivity measure, we advocate developing more comprehensive analysis on cross-ecosystem connectivity comprising at least several main habitat types that are representative of those listed as being of Community Interest in Annex 1 of the Habitats Directive.

Relevant indicators

In the present section we focus on a few promising indicators and do not discuss the many, often relatively simple, ones that have been proposed over the last two decades. For an in-depth overview see Kindlmann and Burel (2008) or the documentation of the Fragstats program (McGarigal et al., 2002). The strengths and weaknesses of several indicators are discussed more in detail in Jaeger (2000) and Pascual-Hortal and Saura (2006) and some suggested improvements are provided in Box 1 below.

Fragmentation of natural and semi-natural areas[7]*.* This is SEBI indicator 13 (EEA, 2009) and includes 3 sub-indices that have been further developed at JRC. Sub-indicator 1 assesses where, how and how much, a piece of natural land (alternatively a piece of forest lands) is surrounded by other "natural" lands (unfragmented), or is intermingled with agricultural and artificial lands. Sub-indicator 1 is based upon a Landscape mosaic model of fragmentation (see previous section for this sub-indicator applied to natural/semi-natural lands). Two further sub-indices are calculated for forests only: core forest fragmentation sub-indicator 2 which measures the intensity of the breaking apart of patches over time, and forest connectivity sub-indicator 3 based on the *Equivalent Connected Area ECA* (see details further) which accounts for patch areas, inter-patch distances, species dispersal probability and landscape permeability. The landscape mosaic model sub-indicator 1 is relevant to inform on the fragmentation pattern of any ecosystem or group of ecosystem for a region of interest in terms of 4 main fragmentation types, type of interface zones and proxies of permeability, and the average patch size for unfragmented pattern. It was calculated for Europe at a broad scale (CORINE Land cover at 25 ha) due to the lack of European-wide data input at fine scale while the forest-connectivity indicators were addressed at both broad and fine scales (spatial resolution of 1 ha). Data derived from CORINE Land Cover data enable to observe fragmentation and connectivity only at broad scale, spatial details below 25 ha minimum mapping unit cannot be 'seen. Moreover, there is the additional problem that the classification used in CORINE is set up with agricultural purposes in mind and very broad for ecological purposes. For European policy purposes, analyses at NUTS levels 3 and 2 seem suitable. Sub-indicator1 is appropriate to describe land fragmentation by agriculture and artificial land, and used as a proxy for land permeability and resilience as soon as data input are fully appropriate. The low spatial resolution and the broad categories in the European wide CLC data reduce the practical applicability of the landscape mosaic indicator to assess ecosystem resilience and we therefore judge this as *moderate.* Increasing the spatial resolution of the input data to the same accuracy level as the forest data (1 ha) and refining the CLC classification for ecological purposes would improve its applicability greatly.

[7] http://www.eea.europa.eu/data-and-maps/indicators/fragmentation-of-natural-and-semi/fragmentation-of-natural-and-semi. ; This indicator itself was developed by the Joint Research Center (JRC) (update on indicator and derived products are at http://forest.ec.europa.eu/forest-pattern and http://efdac.jrc.ec.europa.eu/pattern/map/). It consists of 3 sub-indicators (landscape mosaic, core forest fragmentation, forest connectivity)

Effective mesh size. The indicator is focused on landscape fragmentation by transport networks. It is based on the probability that two organisms at different localities in the landscape can reach each other without having to cross barriers like roads or railways. Multiplying this probability by the total area yields the *effective mesh size* (Jaeger, 2000). The fact that this indicator is one single value that is easy to understand is considered a large advantage by many users (EEA & FOEN, 2011). It has obvious advantages in identifying constraints for wildlife movement and is used especially in the Alpine region but recently also for Europe as a whole based on CORINE Land cover data and transport vector data (EEA & FOEN, 2011). A closer look at the results of the latter exercise shows a limitation of the approach: highly-intensive large-scale agricultural areas tend to have larger mesh sizes than semi-natural landscapes, simply because parcel size in the latter is smaller and thus road density higher. This points to a second, more or less related characteristic of this indicator in that it does not take the *quality* of the matrix into consideration. By doing so its usefulness would be significantly improved. A second improvement would be to adopt the parameter for other scales in order to analyse scenarios for different types of organisms. This would lead to different mesh sizes for different organism types (EEA & FOEN, 2011) but we do not consider that as problematic. In its present state we assess the applicability of this indicator to assess ecosystem resilience at best as *moderate* but promising improvements are possible.

Integral Index of Connectivity IIC and *Equivalent Connected Area ECA.* Both indicators stem from the same family of models. They account for the intra-patch connectivity, the inter-patch distances, the species dispersal capability and can also account for the landscape permeability between focal patches. The probability of connections between habitat patches depends on the dispersal distance of the species and is addressed within a landscape unit. An important advantage of these indices is that a patch is no longer necessarily homogeneous but a figure for the proportion of suitable habitat within a patch can be included, thus yielding more realistic figures. IIC is a binary indicator based on an arbitrary threshold below which patches are considered connected, whereas ECA has a continuous distribution (based on probabilities). The latter is much more realistic but has also a higher data and computational demand. ECA is defined as "the size of a single habitat patch (maximally connected) that would provide the same value of the probability of connectivity as the actual habitat pattern in the landscape" (Saura et al, 2011).

ECA has been slightly amended and applied recently to assess the connectivity of European forests in 2006 and trends in 1990-2000-2006 therein (Saura et al, 2011; Forest Europe, UNECE & FAO, 2011) and shows very promising results. The results become especially meaningful when changes in ECA are compared with changes in total forest area or in relation to changes in matrix quality. The first comparison enables impacts due to changes in *connectivity* and those due to changes in *area* to be distinguished, the second distinguishes between impacts of connectivity changes and those due to changes in overall matrix quality (Saura et al, 2011). Saura et al (2011) and Estreguil and Caudullo in Forest Europe, UNECE & FAO, 2011 used a simple approach when accounting for the matrix quality in the European-wide application to reduce computational demand: the latter assumed the landscape homogeneous between forest patches while the former provided an average per region for the change of matrix permeability but this can be relatively easily refined by

incorporating more realistic parameters derived from least-cost modelling (e.g. Adriaensen *et al*, 2003; Larue, 2008; Estreguil and Caudullo, 2010) or modelling techniques based on circuit theory (Mcrae and Beier, 2007). European-wide assessments require suitable data input to the indicator model, such as land cover maps at appropriate scale. The currently available land cover maps thus provide a broad scale observation of connectivity (CLC at 25 ha minimum mapping unit) and a finer scale observation for forest connectivity only (European-wide forest type map for the year 2006 at a resolution of approximately 1 ha). All estimates are based on assumptions regarding dispersal probabilities of one or more typical species, whereas this knowledge is largely absent for most species and should be further developed by species eco-profiles groups. This may limit its usefulness for real-life applications but is much less of a problem when comparing large-scale policy scenarios.
We estimate the applicability of this indicator to assess ecosystem resilience in policy analyses therefore as probably *good* to *very good* with the remark that the indicator still needs thorough evaluation.

Box 1: Potential improvements in connectivity-related indicators

A major improvement in the applicability of connectivity and/or fragmentation-related indicators will be attained when the spatial resolution of the input land (habitat) cover maps is increased. The observation scale of fragmentation and connectivity depends entirely on the scale of the input data into the model. None of the above-mentioned indicators are "bad" in themselves but all are constrained by the quality of the input data for a European-wide harmonized coverage.

A second general improvement can be achieved if the indicators are calculated for relevant main habitats from the Habitat Directive (eg coastal areas, freshwater wetlands, heathland, natural and semi-natural grasslands, mires, woodlands) and then compared. We realise that there are significant problems translating CLC data and other spatial information into habitat categories but we believe that even such incomplete comparison is to be preferred over a situation where indicators are based on only one or at best a very few habitat types for which more accurate data are available. As discussed above the latter approach may lead to misleading conclusions.

Redesigning CLC classes to suit also ecological needs would be a major step forward in increasing the applicability of fragmentation and connectivity indicators. An automatic re-interpretation of the existing Flemish Biological Valuation Map into a map with Natura 2000-habitattypes shows a reasonable classification error, although such approach has certainly its limits.

A last improvement that we advocate is to vary the (spatial) assumptions on which the different metrics are based. For example, small roads or firebreaks are probably not barriers for movement of mobile organisms, and therefore not included in the calculation of effective mesh size, but at the same time they may be unsurpassable for low mobile organisms. Of course in- or exclusion of different types of barriers will result in different figures for effective mesh size but we believe this will result in much more realistic estimation of ecosystem resilience. Other examples are the thresholds and probability functions used in metrics like ECA. Saura et al (2011) evaluated the effects of varying such parameters (1, 5, 10, 25 kilometre) but, unfortunately, did not do so for the ecologically higly relevant distance of 100 metres (or less). Of course the main reason is the spatial resolution of the CLC data. This again points to the necessity of upgrading the spatial resolution of CLC data.

1.6 F. Resilience increases with ecosystem condition

An indicator of ecosystem "well-being" is needed but in practice indicators related to disturbance are more measurable and therefore "good condition" tends to be defined as an absence of (human-induced) disturbances. Obvious indicators in this respect include SEBI indicators for *Critical load Nitrogen exceedance, Invasive Alien Species* and *Freshwater quality.*

However, some indicators do assess the status of important ecosystem attributes. SEBI indicator 18 assesses the amount of deadwood in forests, which is of particular importance as many forest species and processes (eg nutrient recycling) are highly dependent on the presence of adequate standing and fallen deadwood.

Assessments of the conservation status of habitats of Community interest, that are undertaken by Member States in accordance with requirements under the Habitats Directive, should take into account the full range of habitat attributes that define their condition (eg biophysical conditions, vegetation composition and structure, ecosystem processes and status of key species). Such assessments therefore perhaps provide the best means of assessing the overall impacts of Green Infrastructure initiatives and other conservation actions in an integrated manner. However, the assessments only cover threatened habitats in the EU, are only carried out every six years (with only one conducted so far) and tend to be inconsistent in their methods across countries.

<u>SEBI indicators that can be potentially used</u>: The indicators mentioned above provide relevant and available data, and although each is individually good, they only cover a small selection of pressures, or in the case of conservation status assessments under the Habitats Directive, only cover a proportion of the environment. Therefore as a set we assess their usefulness to assess ecosystem resilience as *moderate.*

References

See main report reference list.