European Commission

Technical Guidelines for assessing and monitoring the condition of Annex I habitat types of the Directive 92/43/EEC

Guidelines to assess fragmentation of habitat types

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Contents

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Executive summary

Fragmentation affects the persistence of habitats over time in a way that is not usually measured by methods that assess habitat condition. As a habitat becomes fragmented, the remaining patches will have a reduced capacity to maintain functional exchanges with other patches; have increasing length of edges exposed to interactions with adjacent, possibly invading ecosystems to which they might not be adapted; and may shrink to a size below the minimum required to maintain the species pool and its associated ecological functions and no longer be representative of the habitat type.

While this is true globally, habitat fragmentation is of particular relevance to Europe, where humans have transformed the landscape for thousands of years, with an exponential acceleration after the industrial revolution. Habitat loss in Europe is addressed by the Habitats Directive (HD) 92/43 EC, which protects a specific catalogue of European habitat types (HD Annex I habitats) and mandates the creation and protection of a network of sites where the habitats can be protected, restored and re-established (the Natura 2000 network).

This guide provides some methodological guidelines on spatial aspects of fragmentation that apply to any terrestrial habitat type. It describes methods that are useful for practitioners dealing with habitat condition assessment. The aim is to show how fragmentation can be included in a decision framework for assessing habitat condition, and examples are developed as necessary. This guidance: i) reviews how fragmentation is currently incorporated by EU member states in their protocols for Annex I habitat assessment and monitoring, identifying convergences, gaps and harmonization potential; ii) describes a decision framework that can be used to assess fragmentation as an explicit contribution to the condition assessment of any terrestrial habitat type; and iii) illustrates an approach to such a decision framework, both to make it immediately applicable by EU member states with small data requirements and objective interpretation clues, and to show how other techniques should be incorporated into such a decision framework.

Habitats are naturally fragmented as an effect of their spatial organization in response to environmental conditions. However, fragmentation produced by human pressures on the environment is much more extreme in spatial and temporal scales than natural fragmentation. Human pressures and land use change take away space from natural and semi-natural habitat types, either for agro-ecosystems or for completely artificial surfaces. Human communication networks ensure functional transferences between socio-economical subsystems, often at the expense of isolating habitat patches.

The operational definition used here is that *the fragmentation of a habitat type refers to any deviation in the spatial arrangement of its observed distribution with respect to that of its expected distribution in natural conditions, which can be interpreted in terms of disturbance*. Disturbance refers to any major disruption of the habitat type distribution to which it is not adapted, such as a shift in environmental conditions at catastrophic rates of change. Human land transformation is often the driver, but any fast process, for example rapid climate change, could be a disturbance.

The guidelines for reporting on Annex I habitats under Article 17 of the Habitats Directive do not address fragmentation explicitly and it is not mentioned in the evaluation matrix for conservation status assessment. The guidelines focus on the assessment of the aggregated habitat area rather than its spatial structure, and fragmentation is treated as a pressure rather than a quantified indicator of habitat condition. EU member states use a great variety of approaches to incorporate fragmentation in their reporting, reflecting different national and regional situations rather than any alignment with comparable methods across the EU. This makes it impossible to draw conclusions about general or habitat specific fragmentation patterns and their regional variation. Most of the member state methodologies reviewed for this guidance cannot be standardized and do not align with the landscape level characteristics of the System of Environmental-Economic Accounting (SEEA) ecosystem condition typology (United Nations, 2021). The SEEA method requires that conservation assets are identified for each habitat type in relation to their habitat condition to precisely assess their conservation status. Such assets result from habitat condition, which is partly informed by the mentioned landscape level characteristics.

This guidance defines the requirements for fragmentation metrics, algorithms, tools, input data and reporting of results that are needed for standardization and then harmonization of methodologies to assess fragmentation. We describe an approach that can contribute effectively to determine the condition of habitat types, and that can be applied by any member state in any biogeographic region. This approach should enable the comparison of member state reports to produce an overview of habitat fragmentation across the EU. Any member state developing or revising its own approach should follow similar requirements as those proposed here, so that convergence and harmonization can be achieved across the EU.

The fragmentation assessment method explained here deals with the frequency distribution of patch sizes. It was originally developed for terrestrial habitat types, principally forests and shrublands. However, the essence of this approach may be easily adapted to other habitats where fragment size is relevant, which gives this technique a wide potential for application.

The assumption is that natural patchiness results from the evolutionary traits of the habitat. Patchiness may be summarized as a frequency distribution of patch sizes. In general, large patches are always relatively scarce with respect to small patches. The relative frequency of large and small patches indicates the degree to which the habitat has been fragmented. Patches must have sufficient size to maintain ecologic functions and contribute to the stability and resilience of the whole habitat area. In landscapes with high rates of land use change, fragmentation operates at faster rates than the habitat can adapt to, and the frequency of large patches decreases abruptly, beyond a certain size threshold, which is the minimum size critical to maintaining habitat functions.

The frequency of large patches above the minimum threshold determines the severity of habitat fragmentation. These metrics are used in a decision tree to determine whether the habitat under consideration has natural patchiness, is in an early fragmentation state, or is severely fragmented. If the habitat is severely fragmented, the habitat patches that are smaller than the fragmentation threshold should not be accounted for in the SEEA. In practice this reduces the stock of habitat assets by eliminating the contributions of all patches that do not meet the minimum size needed to maintain habitat functions. This would reduce the habitat area, which would be lower than the result from the observed distribution that counts all the small patches.

Assessing fragmentation in this way automatically classifies the observed distribution into two categories: habitat assets and restoration candidates. The habitat assets category is likely to be very dynamic, as it reflects directly the impact of fragmentation, which may be operating at a fast rate. Therefore, the contribution of this approach to the evaluation matrix can be expected to be responsive. The category of restoration candidates identifies habitat patches that could be upgraded to assets after appropriate restoration.

The described methodology is a plausible mechanism by which fragmentation may have an impact on the parameter habitat area, and how the assessment of habitat condition at the patch level can impact two parameters: area, and structure and function.

This guide recommends that member state reporting under Article 17 of the Habitats Directive should be adapted to incorporate feedback mechanisms articulated under a conservation accountancy framework, preferably harmonized with the UN SEEA. This would have the important implication that quantities and parameters from the six yearly reports would be comparable, directly controlling the central concepts of habitat assets and habitat stocks. This way, the reporting under Article 17 would become proactive with a return period of six years, which reflects the fast changes affecting European habitats.

1. Introduction

1.1 Habitat fragmentation

Habitat fragmentation and habitat loss are major drivers of species extinction (Cushman et al., 2010), especially in Europe, where humans have transformed the landscape over thousands of years, with impacts accelerated enormously after the industrial revolution. Areas of natural habitat have been shrunk, criss-crossed, and perforated to accommodate human infrastructure, industry, forestry, agriculture, and urbanisation, resulting in a highly fragmented landscape mosaic (Fletcher & Fortin, 2018). Patches of natural vegetation often remain only as islands of marginal land embedded in a varying matrix of land uses. In this situation, it is important to question whether these remaining habitats can persist over the long term.

As a habitat becomes fragmented, the remaining patches will have a reduced capacity to maintain functional exchanges with the other areas of habitat; they will have an increasing length of edges exposed to interactions with adjacent ecosystems that may lead to invasions to which they might not be resistant; and the fragments may shrink to a size below the minimum required to maintain the species pool and associated ecological functions. Many other fragmentation effects have been observed in a variety of ecosystems (Fletcher & Fortin, 2018). As the process proceeds, patches will get smaller and farther apart, and will eventually disappear without further pressure just because of internal instability.

As habitat fragments become smaller and more isolated, the loss of ecological functions means these areas cease to be assets that contribute to meeting the conservation objectives of that habitat type and its associated species. These fragments no longer contribute to the overall functional habitat area and hence, the total habitat extent acknowledged to effectively contribute to such objectives should be reduced. Therefore, in addition to the impact fragmentation has on the internal dynamics of the concerned patches, there is also a more general impact on the accounted distribution of the habitat in question in relation to conservation policy.

Habitat loss in Europe is addressed by the Habitats Directive (HD) 92/43 EC (European Commission, 1992). The Directive protects a specific catalogue of European habitat types (HD Annex I habitats) and mandates the creation and protection of a network of sites where the habitats can be preserved, restored and re-established (the Natura 2000 network). The Habitats Directive Articles 3 and 10 refer to habitat connectivity and to additional landscape elements, implying that the spatial protection of ecological networks and connections is a priority in European conservation policy. Although fragmentation is not explicitly mentioned in the Directive, the Commission guidance on Article 17 reporting under the Directive states that '*it is clear that fragmentation can disrupt the functioning of habitats which are not naturally fragmented and that it is a factor that should be taken into account when assessing structure and functions'* (European Commission, 2023).

The implementation of the Directive is supported by guidance, including this document and the linked set of guidelines for the condition assessment and monitoring of the Annex I habitat types. Because the methods used to assess habitat condition in the field do not usually address the relatively gradual impacts of habitat fragmentation, this guidance covers methods that are complementary to the habitat specific guidance.

1.2 Scope and objectives of this guide

Fragmentation has an indirect but significant impact on the condition of a habitat type and is part of the assessment of Annex I habitat structure and functions. Each of the guidelines produced within the project Guidelines for assessing and monitoring the condition of Annex I habitat types of the Directive 92/43/EC include a dedicated section describing how fragmentation affects the Annex I habitat types and some accepted metrics for how to measure it. This guide is complementary to the habitat specific guidelines.

Fragmentation must be addressed at the landscape level even if it refers to particular habitats (Fahrig, 2003; Fletcher & Fortin, 2018). While the effects of fragmentation differ between habitat types, the description of fragmentation is largely geometric and spatial, and the approach is therefore common to all habitat types. This document provides methodological guidance on assessing spatial aspects of fragmentation that apply to a variety of habitat types. The guidance shows how fragmentation can be included in a decision framework to determine habitat condition. It emphasizes methods that can be taken up by practitioners dealing with habitat condition assessment, with illustrative examples.

This guide is not a manual or review on fragmentation from a scientific perspective (though it cites many of the key academic publications on the subject) and does not report on the state of fragmentation in Europe (see European Environment Agency, 2011) or describe initiatives to avert or control landscape fragmentation (see EEA webpages on Green Infrastructure).

This guidance:

i. reviews how fragmentation is currently incorporated by EU member states in their protocols for Annex I habitat assessment and monitoring, identifying convergences, gaps and harmonization potential.
ii. suggests and describes a decision framework that assesses fragmentation as an explicit contribution to the condition assessment of any Annex I habitat type.
iii. illustrates an approach to such a decision framework, both to make it immediately applicable by EU member states with small data requirements and objective interpretation clues, and to show how other techniques should be incorporated into such a decision framework.

Section 1 of this guide describes a conceptual framework explaining fragmentation and related problems to set a common understanding and vocabulary for the rest of the guide.

Section 2 presents an analysis of current EU member state methodologies to characterize fragmentation. It reviews the types of methods, their utility for assessment and monitoring, replicability potential, and points to strengths and gaps in the reviewed methods.

Section 3 deals with harmonisation. It presents the requirements for a harmonised methodology, including the use of free and open data and methods, peer review of a method prior to its uptake, the use of explicit hypotheses, and differences between harmonization and standardization. It describes a method that integrates the proposed requirements, including its application and additional parameter estimates. The method is integrated in a decision workflow to determine the impact of fragmentation on the habitat type, to exemplify the design and application to obtain meaningful results. The aim is to enable the direct application of each technique by Annex I habitat survey practitioners.

Section 4 points to gaps and areas that need further development, such as common data needs, a repository for fragmentation methods, and aggregation of results at increasing management levels.

1.3 Conceptual framework

1.3.1 Habitats, habitat types, and fragmentation

The definition of a habitat encompasses the conditions present in an area that enable the survival and reproduction of specific organisms. Habitat types are defined to provide an umbrella concept that refers to many organisms, including plants and animals, living in a particular environmental situation and configuration of environmental variables. The EU Habitats Directive defines natural habitats as 'terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural'. These Annex I habitat types, known as habitat types of Community Interest refer to vegetation types that correspond to the terminal stages of ecological succession (Krausman & Morrison, 2016) but also include certain vegetation types associated with traditional land management that maintains them in an earlier stage of succession, known as semi-natural habitats. Therefore, the ecological condition and fragmentation of a habitat type of Community Interest will affect the organisms typically associated with it.

Theoretically, a natural habitat type in the absence of pressures and competition would occupy all the suitable space within its range. The amount of available space for a habitat in an area is determined by the amplitude of environmental variables the habitat type is adapted to and can tolerate, and on whether it competes for space with other habitat types that share similar environmental requirements. These environmental constraints create a pattern of **natural fragmentation** that varies in sharpness and texture according to the gradients of the defining environmental variables. This is the habitats' natural long-term pattern, and the organisms associated with a given habitat type have adapted to it with traits that allow them to transfer and migrate between habitat patches, ensuring their persistence over time. Natural fragmentation is therefore an effect of the spatial organization of a given habitat type.

Fragmentation produced by human pressures on the environment is much more extreme in spatial and temporal scales than natural fragmentation, i.e. it occurs faster and divides more. Human pressures and land use change take away space from natural and semi-natural habitat types, either for agri-ecosystems or for completely artificial surfaces. Human communication networks ensure functional transferences between socio-economical subsystems, often at the expense of isolating habitat patches. Most of these transformations used to be local or regional, but the expansion of human activities and globalization extend the impact of land uses well beyond the regional or national level (Martínez-Valderrama et al., 2021). Furthermore, such changes are often both fast and long-lasting, and natural ecosystems have little or no chance to adapt.

Conservation practitioners operate within a densely occupied mosaic landscape where natural habitat types occupy patches of land that are arbitrary both in size and spatial distribution, and which may be suboptimal for them (Hodgson et al., 2011). In this situation it is safe to assume that the remaining patches of a habitat type risk being incomplete subsets of the energy, matter and information networks necessary for any ecosystem to thrive (Jørgensen & Fath, 2004). Therefore, the persistence over time of the whole habitat may be compromised, even if its condition is temporarily favourable. This is the target problem of fragmentation studies.

Fragmentation can be addressed at three progressive levels of detail: whole landscape, habitat types, and habitat connectivity. This guidance focuses on assessment at the level of habitat types, but the following sections describe the advantages and disadvantages of each approach to explain the context.

1.3.2 Landscape fragmentation

Landscape fragmentation can be addressed as a whole, without considering the situation of the habitat types present in the landscape. An analysis of the overall landscape mosaic seeks relationships between its patterns and processes. Often, such relationships become visible as departures from some type of landscape neutral model where defined patterns are disconnected from any causal process, and that therefore works as a null hypothesis (Gardner & Walters, 2002). In such a model, neighbourhood relationships between elementary spatial units (e.g., raster cells) define patches, and higher-level relations between patches define patterns. The application is geometrical, and advanced tools are available to describe a landscape structure, particularly for raster maps depicting categorical landscape patterns, such as the widely used Fragstats (McGarigal, K.; Marks, 1995).

However, this landscape analysis approach has some important restrictions. Many of the landscape pattern metrics used in such models are sensitive to habitat abundance, which limits their usefulness to assess habitat fragmentation. Only a small subset of metrics can distinguish between habitat abundance and fragmentation, and even in this case any optimal selection depends on the underlying biological rationales (Wang et al., 2014).

Another restriction is that the different landscape classes, categories or components should be defined at an equivalent organization level (Gregorio, 2016). In land cover classification systems that set hierarchical classifications, all land cover classes have comparable complexity and are defined with similar detail at any level. Any landscape analysis should be made within the same level, preserving the target classes at their nominal aggregation. If this restriction is not respected and some target classes are expanded to the next level, the result may contain abnormally small patches or even create opposing association trends between classes.

That restriction is compounded by using whole landscape analysis with only a certain theme as the subject, because general classification systems do not provide the necessary detail. For Annex I habitat types, the required level of detail is unlikely to be found in any general land cover product. For instance, the CORINE land cover data set (EEA, 2007) uses three hierarchical levels with 5, 15 and 45 land cover classes respectively, and distinguishes only three types of forests (Broad-leaved, Coniferous and Mixed). This is clearly too coarse when compared with the 87 forest habitat types described in Annex I of the Habitats Directive (European Commission, 2013). This is why studies using CORINE land cover as a basis for assessing forest fragmentation tend to focus on the continuity of generic forest masses (Estreguil et al., 2012, 2013).

Evaluating fragmentation at the landscape level is suitable for measuring the potential movement of larger wildlife across heterogeneous landscapes. The effective mesh density technique has been applied to 28 European countries (European Environment Agency, 2011) and is a candidate for an EU-wide standardized approach, but this approach is too coarse to assess how fragmentation affects the condition of specific habitat types.

1.3.3 Fragmentation of habitat types

The fragmentation of habitat types can be assessed individually while still working at the landscape level. In this approach, fragmentation is an attribute derived from the spatial distribution of a given habitat type in each study area. This guide focusses on this approach.

An operational definition of this approach is that *the fragmentation of a habitat type refers to any deviation in the spatial arrangement of its observed distribution with respect to that of its expected distribution in natural conditions, which can be interpreted in terms of disturbance*. Disturbance refers to any major disruption of the habitat type distribution to which it is not adapted, such as a shift in environmental conditions at catastrophic rates of change. Human land transformation is often the driver, but any fast process, for example rapid climate change, could be a disturbance. This operational definition of fragmentation has the advantage of considering fragmentation as a purely structural aspect of the involved habitats. Functions and processes may (and should) be linked to it, but by not including them in the definition, causal relationships will be clearer.

There are some practical advantages for assessing habitat condition and for conservation management from this approach to fragmentation. Firstly, the availability of many methods and metrics facilitates a flexible approach adapted to the problem under consideration. Fragmentation as a property of structure can be described using spatial methods such as autocorrelation using variograms, patch patterns, patch morphometry, patch size frequency distributions, and many more. It is easy to lose focus on the real problem being analysed with the large number of techniques available to describe fragmentation. It is possible to easily derive many indicators from any specific landscape in the hope that the variety of results somehow characterize its fragmentation. However, it is generally preferable to start with a hypothesis of fragmentation and then aim to verify it with a limited set of indicators. Whilst this seems obvious, this approach is less frequently used than expected, especially in land management contexts. This point is clarified in Section 3 of this guide.

A second advantage is that habitat fragmentation metrics must be validated, usually in terms of statistical significance, but their interpretation is objective and applies directly to the problem. This is illustrated with a hypothetical example: A certain habitat type distributes regularly in a study area in natural conditions, but economic development progressively fragments the original distribution. The habitat type starts losing extent by change in its spatial distribution pattern. Initially, most habitat loss will take place in areas where land use change is concentrated, causing the distribution to convert from regular (uniform) to contiguous (clusters of patches). As land use change proceeds, habitat patches will only occupy spots that were left arbitrarily apart during the landscape transformation, and the distribution becomes random (scattered patches). A conservation practitioner decides to monitor the process using the Moran's I (Moran, 1950), an autocorrelation metric ranging from -1 (regular) through 0 (random) through 1 (autocorrelation). The practitioner transforms the I raw values to z-scores so that significance can be attached to the results. In other words, the metric is fitted to the hypothesized problem and then statistically validated to yield unambiguous results that can be interpreted directly in relation to the ongoing process. The Moran's I is an absolute metric in the sense that it can be interpreted by itself and does not need to be compared to reference cases placed elsewhere. It can be used to build a trend from assessments repeated over time. The habitat fragmentation approach is therefore very useful to infer risk levels of persistence of habitat types and to generate early warning alerts.

1.3.4 Connectivity of habitat types

Going beyond the structural analysis of the habitat fragmentation approach to landscape connectivity requires the introduction of a process component that represents some movement or transference across the landscape. *Connectivity refers to the capacity of a given entity (e.g., habitat type, species) to make functional transferences between its dispersed patches across a heterogeneous landscape*. Connectivity is therefore a property of the landscape but is defined specifically for the entity (individuals, genes, energy or matter) being analysed.

Ecological connectivity has gained considerable interest as habitat and biodiversity loss increased, and it is the paradigm behind protected area networks such as Natura 2000, where the conservation emphasis is on the functionality of the complete system rather than on self-contained reserves. Unlike in other regions of the world, such as Australia or North America, most protected areas in Europe are relatively small pieces of land that were designated opportunistically on marginal land. Individual designated sites cannot be expected to buffer fluctuations of the target species or habitat types relying only on the resources located within their boundaries. In the Natura 2000 network, the assembly of multiple sites, or ecological network (Bennet, 2004), should be designated in a way that provides transference mechanisms that compensate fluctuations in each individual site, and where ecological connectivity plays a main role in conservation.

Connectivity is a term loosely applied to a variety of processes in the landscape, including demography, population dynamics, migration, genetics and even hydrology. Conventionally, a distinction is made between structural and functional connectivity. In ecology, the former refers to the spatial structure of the habitat for which connectivity is being evaluated, while the latter refers to the process of movement or dispersal across the fragmented landscape matrix.

It is important to note that both types of connectivity must be defined for a specific entity. The fragmentation described by structural connectivity is therefore specific to that entity, in contrast to the more general concept of fragmentation addressed in the previous section. For example, the fragmentation of a certain forest type can be described in absolute terms using statistical metrics, but how such fragmentation will affect the connectivity of that landscape for a certain species requires a specific study. In other words, the same forest fragmentation will have different effects on amphibians compared to passerine birds or large mammals.

Ecological connectivity methods fall into three categories (Fletcher & Fortin, 2018) described below.

Purely structural connectivity is often approached by measuring habitat patch continuity considering the whole landscape. In many cases this analysis is based on the pioneering work of Forman (1995) and uses proximity metrics and their implementations in tools such as Fragstats (McGarigal & Marks, 1995). This methodology has relatively few data or parameterisation requirements, and for that reason it is still widely used.

The second approach codifies the landscape according to the differential permeability it opposes to the transit of the entity under consideration. Such permeability, also called friction or resistance, may be parameterised based on expert knowledge, as done for example to identify ecological corridors based on forest affinity (WWF Spain, 2018) or empirically derived, for example, by taking the inverse of suitability values associated with a predictive distribution, as done to detect zones of maximum connectivity for forest habitat types (Marquez Barraso et al., 2015). This family of approaches requires the definition of origins and destinations for transits, for example using a known observed distribution. After that, ecological corridors and zones of differential connectivity are derived by finding paths of least cumulative resistance, also called least cost paths (Zeller et al., 2012). This approach yields explicit maps of connectivity that can be interpreted in view of the life history of the entity in question. Methodological implementations build on electrical circuit theory to evaluate connectivity, and include Circuitscape (Dickson et al., 2019) and the related package Linkage Mapper (https://linkagemapper.org/). Their conceptual framework implements resistance with great detail but, thinking in terms of Ohm's Law, difference of potential (voltage) has received comparatively less attention.

The third family of methods makes an abstraction of the landscape and presents the connectivity problem as a mathematical graph, which often represents subpopulations as nodes in a meta-population scheme. Such a graph represents the connectivity network, and metrics of size, modularity or connectivity are derived using graph theory (Urban et al., 2009). A tool implementing graph analysis for general connectivity purposes is Conefor Sensinode (Saura & Torne, 2009).

This graph approach is complementary to the least cost path because it is difficult to use metrics describing the connectivity scenarios deriving from the least cost path approach, whilst metrics describing graph structure and functions are readily available from the associated theory. The parameterization of the graph may be a weak point where strong assumptions may be made, while parameterizing resistance is often objective and explicit.

This overview shows that connectivity methods and their results are not always consistent and comparable. What landscape representation is used as input data is strongly dependent on the entity being analysed, as well as the internal model assumptions and the resulting connectivity metrics. Most of the described models deal with potential connectivity, whilst there are few applications that measure actual connectivity, usually represented by animals that can be tracked. Habitat types as target entities of connectivity pose important challenges concerning validation, which probably should be investigated as long-term patterns in the domain of landscape genetics (Baguette et al., 2013; Galpern et al., 2012).

In summary, there are important issues concerning the consistency of methods for assessing connectivity and their validation. Using any model of connectivity often requires the careful and specific use of contributions and empirical parameterisations for the entity under consideration. A species or habitat type can only be properly evaluated under different conservation scenarios using a connectivity study. However, this activity is closer to a research project than to a set of standardized metrics routinely deployed for the assessment and monitoring of habitat types, and is out of the scope of this guide.

2. Analysis of existing methodologies for the assessment and monitoring of habitat condition

Fragmentation, in its wider meaning, is a key concern for habitat conservation, and is included, to some extent, in some of the methodologies for assessing habitat condition applied by the EU member states. This section reviews the implementation of this concept in the member states analysed for this guidance.

2.1 Article 17 of the Habitats Directive

The guidelines for reporting on Annex I habitats under Article 17 of the Habitats Directive acknowledge that fragmentation can disrupt the functioning of habitats which are not naturally fragmented and mention it as a factor that should be considered when assessing structure and functions (European Commission, 2023). However, the EU guidance does not provide any indication on how this could or should be done and does not mention fragmentation in the evaluation matrix. The focus is on reporting on aggregated areas rather than on their spatial structure.

Because fragmentation affects the observed distribution of a habitat type, it affects the range and area at the biogeographical level. Range refers to the geographic envelope where a species or habitat type may be found, while area refers to the surface effectively occupied by that species or habitat within the range. However, the EU guidance requests both range and area to be reported as totals at the biogeographical region level and does not distinguish between fragmented or continuous distributions.

In the guidance, fragmentation is treated as a complement to other attributes and/or a result of pressures. For example, for the conservation status of an Annex I habitat to be considered 'favourable', fragmentation should not impact significantly on ecological processes. Similarly, it is recommended to consider fragmentation when setting the favourable reference area of a habitat type. To evaluate the conservation status of a species' habitat, the guidelines recommend considering fragmentation for its potential to have a negative impact on the population dynamics of the species. The guidance does not request specific estimates or ecological thresholds to support those assessments.

Fragmentation can be reported as a pressure or threat, with pressures acting in the present and threats in the foreseeable future reporting periods. Up to 20 pressures may be listed for each habitat, and the list of possible pressures include land management practices with an impact on fragmentation, for example 'conversion into agricultural land', or 'intensive grazing or overgrazing by livestock'. The impact of such pressures on the habitat area must be specified in terms of scope (proportion of affected area) and influence (on area or habitat condition). The consideration of fragmentation as a pressure indirectly derived from management lends better to a narrative than to quantified indicators. This may be the reason behind the great variety of approaches with which member states incorporate fragmentation in their reporting.

2.2 Member states methods

Fragmentation is considered in very different ways in the methodologies applied by member states to assess habitat condition. For this guidance, we reviewed approaches to include fragmentation in the reporting of habitat condition in some member states that have developed this concept. The methods are very diverse and respond to the perceived environmental drivers in each country, therefore there are not many shared approaches between member states. This section offers an overview of these methods followed by a discussion.

2.2.1. Overview

Austria

In general, fragmentation is addressed as an impairment created by hard linear infrastructures (Ellmauer et al., 2020). It is measured as the proportion of area affected by such facilities, counting roads with a width of 2 m or more and specifying dissection levels if necessary. Certain habitat types are particularly targeted in these terms, for example: 4080 Subarctic willow scrub, 7110 active raised bogs, 7120 degraded raised bogs that are still capable of renaturation, 8150 silicious screes of the mountains of Central Europe, and 91D0 bog forests.

Belgium-Flanders

The methodology used to assess the local condition of Annex I habitat types in Flanders explicitly addresses the role of area in fragmentation (Oosterlynck et al., 2020). The method evaluates the connectivity and patch size in a dedicated section of the evaluation using a GIS approach combined with threshold values and a set of rules that decide on habitat patch and cluster sizes. The central feature are functional habitat clusters, which group related biotope types hosting habitat requirements for their typical species and separated by a distance that is bridgeable by these species. Such resulting clusters are basically landscape types: silt, coastal dunes, waters, dry and wet heathland habitats, grasslands, peat bogs, forests (including dune forest). Two conservation status classes are allocated to them according to the proportion of faunal species that can be potentially present, and threshold reference areas are provided. The spatial cohesion of functional habitat clusters is then measured through the area occupied by them and the distance between similar clusters.

Bulgaria

In Bulgaria (Zingstra et al., 2009), fragmentation is included in the assessment matrix of species, and it is considered as a parameter of habitat structure and functions. Fragmentation is considered mainly in terms of hard barriers to dispersal of target species. The method provides several structural thresholds related to the frequency of fragmenting structures in the monitored polygons. Fragmented habitats are characterized as patches of suitable habitats with size less than 40 km^2, and a contact zone with neighbouring suitable habitats under 50% of the length of their outward perimeter.

Fragmentation of habitat types 1130 'estuaries' and 1140 'Mudflats and sandflats not covered by seawater at low tide' is considered in terms of mapped anthropogenic structures interfering with the boundaries of the habitat polygons. Habitat condition is assessed for fragmentation as the proportion of such human infrastructures in the habitat polygons.

Czech Republic

Lustyk (2023) refers to heterogeneous mixes of vegetation and habitat types that must appear as 'mosaics' in thematic maps. However, there is no further mention of spatial or fragmentation aspects.

France

The method includes fragmentation indicators for forests, grasslands and marine habitats. In the case of forests, fragmentation and connectivity are framed in biogeographical and landscape ecology theory (Maciejewski, 2016), but no concrete indications are given except that measurements must be taken on the site. The most recent method (Maciejewski & Bonhême, 2024) suggests two indicators: fragmentation within-site and connectivity between forest spaces. Fragmentation inside the site is computed in terms of density of linear elements (cumulative length) per hectare. This indicator is also suggested for monitoring. Fragmentation in the environment (connectivity) is assessed for the entire forest area, not for individual forest types in the forest matrix. The connectivity of the forest masses is evaluated in terms of proximity of the assessed forest to silvo-ecoregions.

A full procedure for fragmentation of grasslands has not yet been implemented (Maciejewski et al. 2015). A tentative choice was made of indicators targeting at connectivity in terms of proximity between patches, using Fragstats (McGarigal & Marks, 1995) , for example the Effective mesh size index to express probability of randomly selected points to be connected. Monitoring of habitats is suggested by three methods: visual interpretation, average distance between two habitat patches and spot hierarchy based on Conefor Sensinode (Saura &Torne, 2009). Monitoring the shape of habitats is proposed on the hypothesis that circular shapes are more stable. Also included is the evolution of the degree of connectivity of habitats, which evaluates actual versus potential connections.

Hungary

The sampling protocols for grassland and some wetland habitats require information to describe habitat patches in the landscape mosaic (Varga et al., 2021. The surveyor must report for a given habitat patch: other typical formations at the edges (shrubs, weeds, marsh, etc.); neighbouring habitats (those listed formations that cover at least 5% of the edge); isolation (yes, no, marginal); fragmentation pattern in the landscape (large spotted, rare small spotted, frequent small spotted, rare linear, frequent linear).

Italy

The Italian method proposes an analysis of landscape metrics concerning the shape, composition and arrangement of polygons from landscape ecology theory (Angelini et al., 2016). Several indices are mentioned as examples, such as S (total area for each class), TM (size of the largest patch for each class), Tm (average size of the patches for each class), Nt (number of patches for each class), A/P (average ratio between the surface and the perimeter of the patches for each class). The text elaborates on the potential of the results: "*The spatialization of this information through GIS applications will allow an assessment of the diversity and naturalness of the habitats in a range of scales ranging from fine to large. In this way, basic concepts of landscape ecology, such as isolation, connectivity and fragmentation of habitats will be quantifiable*." However, despite the reliance on field sampling for the proposed landscape metrics, the method does not make explicit the link between the field observations and the polygon metrics approach.

Luxembourg

Fragmentation is only considered as an impairment caused by linear, man-made infrastructures in forest biotopes (Ministère de Developpémenet durable et des Infrastrctures, 2020). No metrics is provided for its assessment or monitoring. The guidance sets a minimum threshold for mapping forest habitat patches of 0.3 ha, and states that patches below that size should not be mapped.

The Netherlands

The method includes detailed landscape level indicators concerning fragmentation and spatial aspects and requires the determination of spatial conditions for a variety of habitat types, including rivers and wetlands. The indicators mostly address barriers to connectivity in terms of the effect of buildings or other infrastructures on the ability of fauna to transit between habitat patches (BIJ12, 2021).

Poland

The Polish method requires fragmentation as an attribute to describe habitat condition, structure and function, and a description of the isolation of habitat types, particularly of grassland habitat types (Mróz, 2010). No specific methods or criteria are given, except a request to address the size of individual patches.

Slovenia

The Slovenian method measures habitat patches with two indicators (University of Ljubljana, 2018). One is the average area of a habitat type patch, according to the hypothesis that the bigger the patch, the better its resilience. The other is the edge of habitat type bordering with other habitat types or with human managed land. The treatment is mostly narrative and does not provide methodology or thresholds.

Spain

The Spanish method considers fragmentation of shrublands and forests through two approaches: functional and structural (del Barrio et al., 2019). Functional fragmentation refers to the resistance a habitat may find to disperse across its surroundings. The resistance is proxied by a measure of the resemblance between the inside and the outside of a target patch, and is not defined specifically for habitat types except the ones mentioned below. The method provided is based on computing the Orloci Chord dissimilarity index, using frequencies of either land condition states of ecological maturity (an independent product used in Spain to assess land degradation, as described in Sanjuán et al. 2014), or land cover classes from the Corine Land Cover dataset. Functional fragmentation computed this way aims at delineating the space where specific projects of connectivity should be developed, and to estimate the degree of isolation of habitat patches.

Structural fragmentation in the Spanish scheme refers to the frequency distribution of sizes of habitat patches, on the basis that anomalous abundance of small patches hampers ecosystem persistence on the long term. A method is provided that uses a multifractal approach to detect significantly truncated frequency distributions.

For riparian forest habitat types, the longitudinal continuity of riparian galleries is taken as a proxy of connectivity between the habitat patches (Lara et al., 2019). Fragmentation of coastal dune systems is estimated in terms of runoff erosion and occupied area (Aranda et al., 2019).

2.3 Discussion

There is a wide variety of approaches to fragmentation in the member state methods reviewed here. It results in general from ad hoc efforts to reflect the environmental situation identified in each country, rather than from alignment with standard methods that could be compared across the EU. This is because, to date, no such need has been articulated. For this reason, the choice of indicators tends to be exclusive and monothematic so that, in general, no member states have chosen to use a range of fragmentation indicators.

Several methods, for example, point to awareness of the impact of linear human-made structures, and record details on the width, length and density of roads. In these cases, patch size and mosaic properties are loosely required or not specified at all. In other member states the target for fragmentation assessment is the spatial structure of habitats, either clustered or for individual sites. We can distinguish between methods where the objective is to evaluate connectivity between patches ('bridging distances') and those that draw information on fragmentation from the frequency distribution of patch sizes. Where the emphasis is on patch size and abundance, less importance is attributed to the impact of man-made structures, especially roads. This is perhaps because the impact of such structures is assumed to be implicit in the resulting patch sizes. There are a few methods that mention landscape ecology in academic terms, giving several indices as examples. However, the methods do not explain how they should be used in Annex I habitat monitoring, and give the impression that heterogeneous indicators have been put together without clear connections with conservation targets.

The outcome of these tailored approaches to fragmentation is that member states focus primarily on the spatial aspects of the impacts on habitat condition. The required monitoring consists of as many aspects as deemed necessary of the spatial arrangement of the Annex I habitat or habitats. In most cases fragmentation is expressed using quantitative or ordinal measures. However, their interpretation is strongly based on expert judgements with no reference scales. This makes it difficult to obtain reproducible results, which is only partially compensated by controls at different stages of methodology application.

These approaches work for national reporting objectives, but the heterogeneous situation across the EU makes it impossible to draw conclusions about general fragmentation patterns and their regional variations, both in general and for specific Annex I habitat types. Most of the member state methodologies reviewed for this guidance cannot be standardized and they do not align with the landscape level characteristics of the System of Environmental-Economic Accounting (SEEA) ecosystem condition typology (United Nations, 2021). The SEEA method requires that conservation assets are identified for each habitat type in relation to their habitat condition to precisely assess their conservation status.

3. Guidance for the harmonisation of methodologies for assessing and monitoring habitat fragmentation

3.1 General considerations

The review of member state approaches to survey fragmentation reveals the absence of a unified scope as well as, probably, of a consolidated theoretical basis. This is not necessarily a bad thing, as many of the approaches are recent and this field is still undergoing scientific exploration. However, it makes it difficult to select a consensus method and the results lose application rapidly beyond the limits of the concerned member states.

The lack of reference values for most of the metrics is striking, especially compared with other ecological fields. For instance, relationships between number of individuals, reproductive rate or carrying capacity are well known in population dynamics. Equivalently, relationships between gross and net primary production and their interaction with total biomass are known in ecological succession theory. For these relationships and other, it is possible to measure parameters in the field and enter them into a model to predict ecosystem responses.

Most fragmentation studies do not specify those relationships but assume that fragmentation has increased with the human exploitation of the landscape and is, therefore, undesirable. Notwithstanding the absence of precise references, this is a dangerous assumption which might backfire by creating unexpected results, for example, that a certain population becomes extinct after demographic disturbances are synchronized in a system of maximum connectivity (Earn et al., 2000).

As a result, the uptake of practical fragmentation techniques by public administrations dealing with habitats management and assessment is still relatively low. This is not because of absence of demand. On the contrary, the evident fragmentation of habitats and ecosystems has created a justified societal concern, which is reflected in many initiatives and calls demanding more assessment and restoration. A key problem is that most of the available techniques require data that must be recorded with targeted and dedicated field effort. Further, many fragmentation results are subject to validation issues that may create a difficult situation if important economical investments depend on them.

The guidelines defined here address the demand for a fragmentation approach that can contribute effectively to determine the condition of Annex I habitat types, and that can be applied by member states whatever the biogeographic region. The approach should enable the combination of member state results to produce an overview of fragmentation across the EU.

Two remarks concerning the guidance in this section: First, committing directly to the harmonization of the existing methodologies is probably not realistic. Harmonization involves converging to common responses from diverse approaches. The review of methods applied by the member states shows that they aim to generate immediate responses to the need to assess the condition of Annex I habitats in their particular context, but the potential to aggregate or generalize results to the biogeographical region level or wider is small. Therefore, some previous degree of standardization should be applied to any method that is candidate to a later harmonization. This means essentially that such method should be applied uniformly by different member states.

Secondly, the approach suggested here is an example to illustrate as explicitly as possible the application of a fragmentation method that may be useful both for individual member states and for higher organization levels within the EU. The real message is that any member state developing its own approach should observe similar requirements as those posed here, making convergence and harmonization possible on the short term.

3.2 Requirements and constrains

Any procedure to assess fragmentation should meet the following requirements:

- The fragmentation metrics should:
  - be intuitive and directly based on observed data;
  - carry an implicit validation (e.g., statistical significance) that enables a direct interpretation;
  - have an explicit numerical scale;
  - have reference values to compare the obtained results to.
- The algorithm should:
  - neatly separate computational from interpretative (e.g., expert judgement) workflows;
  - operate with parsimony, by selecting simple and direct solutions.
- The tool must:
  - be specific about concrete questions, avoiding the production of generic information that is difficult to interpret directly;
  - maximize the quality of results in relation to the effort invested in its application;
  - be updatable and upgradeable;
  - be public, open and free, therefore enabling its replication and improvement by independent users;
  - operate standard methods with a potential for harmonization.
- The input data should:
  - come exclusively from public geospatial databases with defined standards for data quality and updating.
- The results must:
  - reflect the direct effects of fragmentation on present and / or future habitat condition;
  - be replicable beyond any expert intervention;
  - have an unambiguous link with the Article 17 evaluation matrix for conservation status assessment;
  - have a link to environmental policy.

3.3 Analysis of patch size fragmentation

3.3.1 Rationale

Almost by definition, fragmentation involves the division of a previously continuous extent into patches. Thus, its assessment may be more effective by focusing on the patches themselves than by targeting the heterogeneous causes of the division. Moreover, the reason why excessive patchiness threatens a habitat is because it needs a minimum area to maintain its ecological functions, including exchanges with other surrounding ecosystems. Patches below this minimum area may have a reduced or compromised persistence on the mid- to long-term even if the habitat in question is currently healthy (Hastings et al., 1982).

The fragmentation assessment method explained here therefore deals with the frequency distribution of patch sizes. It is possible to use other geometric descriptors such as those described in Section 1.3.2 (perimeter to area relationship, etc.), but shapes are very sensitive to spatial scale and mapping errors, because of the problem of delineating limits between adjacent ecosystems (Levin, 1992). In contrast, area size is a more robust estimate, and this is why it will be used here.

The method was originally developed for terrestrial habitat types, principally forests and shrublands. However, the essence of the approach may be easily adapted to other ecosystems where fragment size is relevant, so this technique can be applied to all terrestrial habitats.

Terrestrial habitat types have a natural patchiness. It results both from the occupation of areas within the amplitude of environmental variables that the habitat type is adapted to and can tolerate, and from biotic interactions (mostly competition) with other natural habitats that might occupy those areas. The assumption of this method is that such natural patchiness results from evolutionary traits, as an adaptation of functional connections within the overall distribution and as a balance between genetic diversity and vulnerability to disturbance propagation (Collinge, 1996).

Patchiness may be summarized as a frequency distribution of patch sizes. In general, large patches are always relatively scarce with respect to small patches. The relative frequency of large and small patches indicates the degree to which the habitat has been fragmented. In natural conditions, this frequency change from few large to many small patches is relatively regular because the habitat adapts to the patchiness drivers and retains its integrity. But in landscapes with high rates of land use change, fragmentation operates at much faster rates than the habitat can adapt to, and there is an abrupt decrease in the frequency of large patches beyond the size threshold critical to maintaining habitat functions (del Barrio et al., 2021). Therefore, departures from that regularity of patch pattern indicate fragmented ecosystems, and the patch size threshold that marks the sharp transition from few large patches to many small patches is a suitable metric.

3.3.2 Overall implementation

This section describes the main assessment workflow. Details of each step are described in following sections.

Patch size frequency analysis is a suitable procedure to detect significant fragmentation. However, using it without further ecological context can lead to reactive and diffuse management responses that do not identify and reduce the fragmentation causes. To take more proactive decisions in a conservation framework, it would be necessary to estimate the actual effect that fragmentation has on a specific piece of landscape and how it affects the condition of the habitat type under consideration. The workflow needs to be enhanced by comparing the fragmentation metrics with other relevant metrics, introducing two important concepts: minimum area threshold and ecosystem assets.

Minimum area threshold (MAT) refers to the smallest patch size that can sustain the basic ecologic functions of a given habitat type in each geographic location. This spatial indicator should be distinguished from the range and area covered by the habitat type, which refer to totals at the biogeographical region level (range being the outer limits of the area effectively occupied by the habitat). In contrast, MAT is defined at a much higher scale as a patch level measure, defined here to flag the size below which a patch of a certain habitat type can no longer be viable on the long term.

In the framework of the SEEA, ecosystem accounting involves recording over an accounting period the stock and change in stock of each ecosystem asset. In turn, each spatial area of a specific ecosystem type is, for accounting purposes, treated as an ecosystem asset. The amount of ecosystem assets may change over time due to ecosystem enhancement and degradation during the reporting period.

Ecosystem accounting has the potential to formalize a dynamic component of the Annex I habitat evaluation performed under Article 17, as pieces of a certain habitat type must overcome some condition threshold to be attributed a certain conservation status. In turn, a minimum conservation status should be achieved for a given piece of habitat type to be considered as an asset. This is the essence of the workflow proposed here (Figure 1).

The most straightforward approach to obtain the variable *Area covered by the habitat type* might seem to derive it directly from the habitat *observed distribution*. However, this is an aggregated total that ignores any aspect of the habitat spatial structure. A habitat type with a natural patchiness in a pristine state can have the same total surface area as a highly fragmented habitat arrangement in an intensively exploited landscape. The highly fragmented habitat could even surpass the *Favourable Reference Area*, leading to an overestimation of conservation status. Thus, it is not recommended to rely on total observed habitat distribution unless there are major issues of data quality.

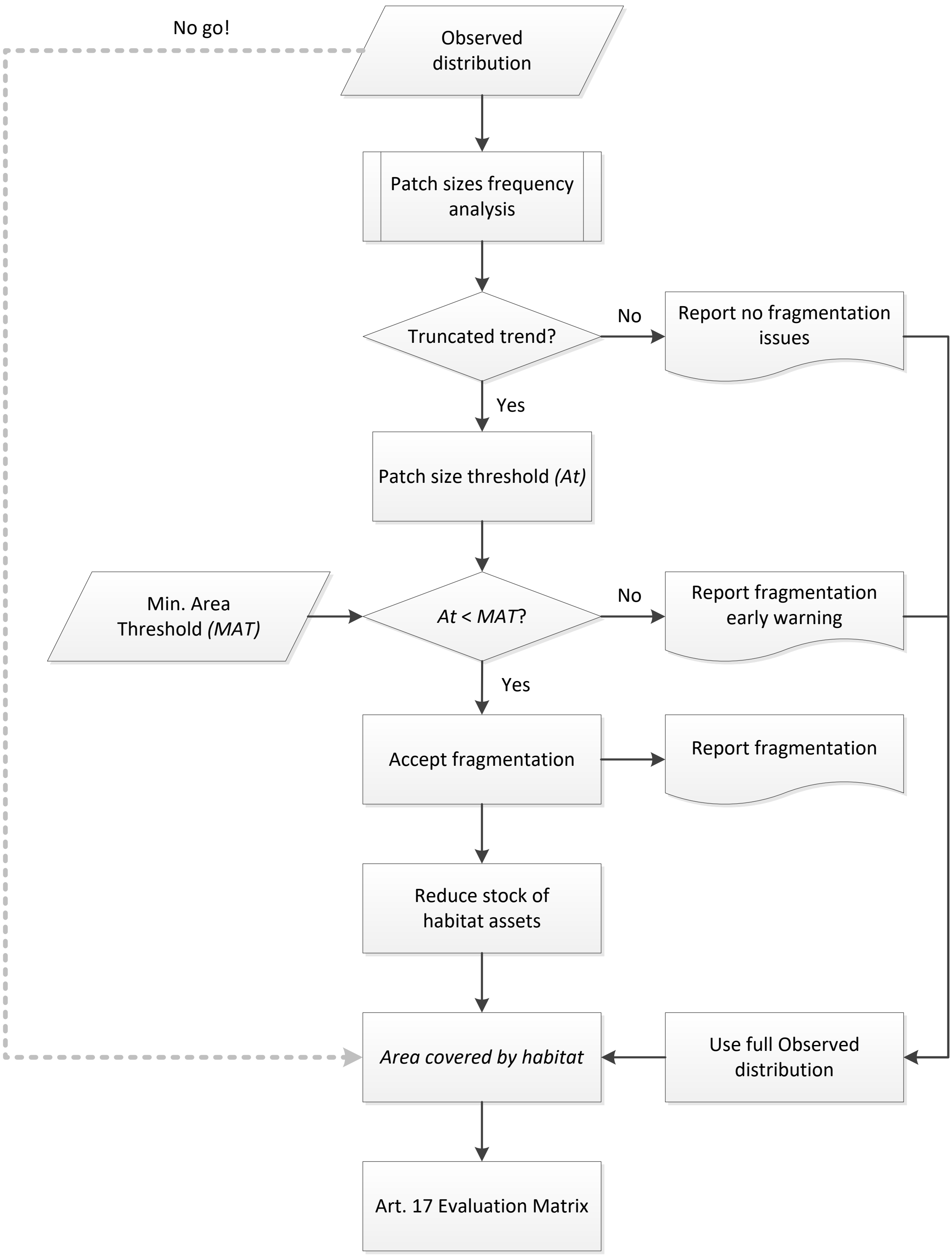
Several member states carry out Annex I habitat mapping at spatial scales of 1: 50,000 or higher, which enables an analysis of the texture of the observed distribution. This could be submitted to a **patch sizes frequency analysis**, a relatively simple statistical technique that determines whether the frequency of patch sizes decreases steadily as size increases. If the decrease is regular and uniform, it can be assumed that the habitat has a natural patchiness pattern, and there are no fragmentation issues to report. In this case, the use of the aggregated extent of the full observed distribution to approximate the Area covered by the habitat type would be justified.

If, on the contrary, there is a significant sharp drop (truncated trend) in the number of large patch sizes beyond a certain area, fragmentation may be suspected. The interpretation is that some pressure is operating that fragments the habitat extent at a faster rate than the habitat can recover it. A decision should be taken by comparing that patch size threshold (*At*) where the trend truncates with the Minimum Area Threshold (*MAT*).

If *At* is greater than *MAT*, it follows that many of the relatively small patches that are very frequent are still above the threshold of patch persistence. This situation may not represent a deterioration in the present, but it should be considered as a threat on the basis of the detected structural fragmentation (truncated trend), which has a potential to intensify in the future. The action should be to report an early warning of fragmentation and, with some reservations, still use the full observed distribution to estimate the *Area covered by habitat*.

However, if *At* is lower than *MAT*, the conclusion is that there are many small patches, the sizes of which do not reach a minimum to ensure habitat persistence on the short term. This confirms the decision to accept fragmentation, and two procedures derive from this finding. On the one hand, to report true fragmentation for appropriate actions to be taken on tracing the management origin of this pattern and on further investigating its possible impact on ecosystem functions. On the other hand, those patches that are smaller than the minimum size to maintain ecologic functions do not contribute to the whole habitat stability and resilience and should not be included in the habitat account. In practice this reduces the stock of habitat assets by eliminating the contributions of all patches smaller than *At* to the Area covered by habitat, which will result in an effective habitat area that is lower than the total resulting from the observed distribution

Figure 1. Flowchart of fragmentation assessment based on patch size frequencies



It could be argued that *MAT* could make a better lower threshold than *At* to decide which small patches to eliminate from the accounting, as the former is a true ecological variable and the latter is a mere statistical outcome. Patches are only excluded if *At* < *MAT*. By using *At* to set the exclusion threshold, only very small patches will be excluded from the accounting. It is true that some of these very small patches would exist in an unfragmented habitat too, but they would be relatively rare and would be connected to bigger patches around them. In the case of the fragmented habitat where *At* < *MAT* a significant part of the distribution occurs as very small patches that may be presumed to have impaired persistence. This justifies not accounting them as assets. Further, in this case *At* is the lowest size threshold to apply, which results in a conservative decision.

Discarding small habitat patches within an ecosystem accounting scheme may sound a drastic proposition. However, an essential property of the habitat types defined in Annex I of the Habitats Directive is their capacity to provide support to many species of conservation interest and the ecosystems they are part of. Therefore, Annex I habitats exist more for their function than for their description, and this means that they should be assessed following approaches with a functional transcendence, such as that described here.

One advantage of proceeding this way to identify fragmentation is that the observed distribution is automatically classified in two categories: habitat assets and restoration candidates. The habitat assets category is likely to be very dynamic, as it reflects directly the impact of fragmentation, which may be operating at a fast rate. Therefore, the contribution of this approach to the habitat evaluation matrix can be expected to be responsive. The category of restoration candidates identifies habitat patches that could be upgraded to assets after appropriate restoration, completing this way the dynamic feedback in the overall management of the habitat distribution. Further, these patches represent objective targets for restoration projects, both for their identified location and current status and for the ecological state to aim at.

The landscape-ecological system analysis (LESA) pathway may be complementary to the described approach, especially for the category of restoration candidates, to understand the functioning of habitats at the landscape level and better assess the impact of the interaction between fragmentation, land-use change and landscape-ecological processes (e.g., hydrology) and landscape dynamics (Decleer & Bijlsma 2021).

3.3.3 Minimum area threshold

As defined above, minimum area threshold (MAT) refers to the smallest patch size that can sustain over the long term the ecological functions of the habitat type under consideration. This is a necessary reference to decide whether a given patch should or should not be considered as an ecosystem asset.

It is important to note that this concept has an inherent difficulty. Some ecological functions mean interactions with surrounding ecosystems, either positive such as exchange of individuals and species with other nearby patches of the same habitat type, or negative such as exploitation made by other different ecosystems in the vicinity. Those interactions depend on the degree of isolation of the patch under consideration within the regional distribution of the habitat type. Therefore, probably there is not a unique MAT value for a certain habitat type. Rather, a MAT estimate should be obtained for each typical landscape setup to account for the regional variations of that habitat type. In fact, each single patch might have its own MAT reference resulting from its particular arrangement within the local observed distribution, and considering the landscape matrix it is embedded in. This is an important theoretical constraint on the estimation of this parameter and whilst it is not possible to follow it entirely, it is important to take a critical perspective to finding suitable surrogates.

Species to area relationship

This approach draws on the theory of island biogeography (MacArthur & Wilson, 1967), which predicts that the number of species (*S*) increases with the size of the area (*A*) following a power law:

$$S = c \cdot A^z$$

Exponent *z* is the growth rate of *S* as a function of *A*. It depends on the involved taxa and other factors of the specific setup to which the relationship is fitted. The rationale is that larger areas include a higher environmental diversity, which in turn may host more ecological niches and therefore more species.

The power law of the species area relationship has a diminishing growth, i.e. it tapers after an initial steep increase, reflecting the initially fast saturation of species as the area increases. The trend may be used to find an area size that potentially contains an important fraction of all the species present in a particular setup. The species to area curve may also be used to estimate biodiversity loss (percentage of remaining species) as a function of area reduction (proportion of habitat loss). This approach has been used by Spain and other member states to find the Favourable Reference Area (FRA) of Annex I habitats (Camacho, 2024). For example, Fernández Palacios et al. (in press) proposes to apply 25% of the potential distribution extent of 9360 Macaronesian laurel forests *(Laurus, Ocotea)* to estimate the FRA of this habitat type, in view of its species to area curve.

One advantage of the species to area relationship is that it is relatively easy to parameterize for a particular setup. The function can be transformed to linear by applying logarithms:

$$\log S = \log c + z \cdot \log A$$

This linear function can then be easily fitted by recording pairs of values (a, s) in the field under appropriate sampling designs.

The species to area relationship was originally devised to explain ‘big’ numbers, such as the biodiversity found in islands of different sizes. The application of FRA is also intended at small scale (i.e., large areas at low resolution). Using a curve that has been fitted using small scale, coarse-grained spatial data (e.g., full islands), to estimate the number of species associated with very small patches, within the high scale, fine-grained observed distribution of a habitat type at a local level, is probably pushing its utility too far. That curve will have been fitted using values of area that are orders of magnitude larger than those of the distribution patches, and its left side is not likely to be very precise.

However, the fact that the species to area relationship is a power law means that it is scale independent. Therefore, it can be used to explore MAT values at high scales if the curve has been fitted using data commensurable with the problem at hand, which in this case is patch size. In other words, a field survey, possibly following a spatially nested sampling design, would be needed to provide the necessary data.

If the data are available, it is preferable to fit the power law curve through the linear transformation described above rather than using automatic statistical techniques such as piecewise regression to detect area thresholds (Matthews et al., 2014). This is clearly a point in the workflow where expert intervention is required. An informed inspection of the resulting curve should yield a suitable MAT, especially if combined with the criteria described next.

Minimum Dynamic Area and its derivatives

A weakness of the application of the theory of island biogeography for designing conservation areas is that the assumed equilibrium between immigration and extinction is not true in a conservation framework. This is because recolonization from unprotected land outside the protected areas cannot be taken for granted, therefore extinction should be assumed to be the dominant process, and the disturbances causing it should receive appropriate attention. The concept of Minimum Dynamic Area (MDA) was formulated to address this problem (Pickett & Thompson, 1978). MDA is defined as *the smallest area with a natural disturbance regime which maintains internal recolonisation sources and hence minimises extinctions*.

There is not a formal protocol to determine MDA. The procedure requires accounting for the size and recurrence of natural disturbances (fires, etc.) occurring in the target area. These data are then used to design protected areas larger than the maximum disturbance found and that are as self-contained as possible in terms of recolonization sources and diversity of post-disturbance patch ages. Interestingly, a last criterium refers to including independent MDA for each habitat type included in a protected area.

Considering natural disturbances as the main driver to determine MDA suits relatively wild landscapes where disturbances are rare and confined. Where this is the case, it may be worth examining the upgraded concept of Minimum Dynamic Reserves (MDR) (Leroux et al., 2007), which makes spatial simulations of disturbance and recolonization to iteratively achieve a stable solution.

However, the MAT notion is concerned with individual patches of European habitat types, which, instead of (or in addition to) having confined natural disturbances, may be themselves embedded in a large disturbance in the form of land use and land use change. A straight application of the MDA or MDR approach would be likely to result in absurdly large values.

Notwithstanding the original scope of MDA/MDR for designing protected areas for more pristine regions than found in Europe, the method has a potential application to the MAT problem. The stress on the dominance of extinction and effects of disturbances (including man-induced ones) on isolated patches is a sound basis to evaluate patch sizes resulting from a more empirical method, such as the species to area relationship.

3.3.4 Patch sizes frequency analysis

Theoretical and practical details of the patch size frequency analysis described here can be found in del Barrio et al. (2021). Sample data to work with along with R scripts implementing the analysis are described in Sánchez de Dios et al. (2021) and can be downloaded from Sánchez de Dios et al. (2020). Therefore, only an overview is offered here to help a prospective practitioner decide whether this technique could be suitable. The text below summarizes the studies listed here, so there are no further citations.

The reference trend for this analysis is set by the fact that many events in nature have an inverse relationship between their size and their frequency. This is true for islands, lakes, earthquakes and habitat patches. It is intuitive that, in general, large instances of any of those examples are much less frequent than their small counterparts.

This was empirically formulated in 1938 as a power law by the Czech geographer Jaromir Korcak:

$$N(a) = c \cdot a^{-B}$$

Where $N(a)$ is the number of events with a size larger than a, c is a constant and B is an exponent with a small range of variation for the type of event under consideration.

As with other power laws, parameterizing Korcak's function is easy by transforming it to a linear function:

$$\log_e N(a) = \log_e c - B \cdot \log_e a$$

Therefore, it is only necessary to sample value pairs (a, $n(a)$) and then fit a liner function to their logarithmic transforms.

As said above, the power law function fitted to the experimental data establishes the reference for undisturbed conditions, in this case, of patch size frequencies. Departures from it can be interpreted as proportional to fragmentation. Such departures can be detected upon inspection of the residuals. If this inspection suggests that the fit could be improved by subdividing the data in two or more intervals of a, a segmented regression with single breakpoint is fitted, and the breakpoint is used as the indicator of patch size threshold (*At*) as described in Section 3.3.2.

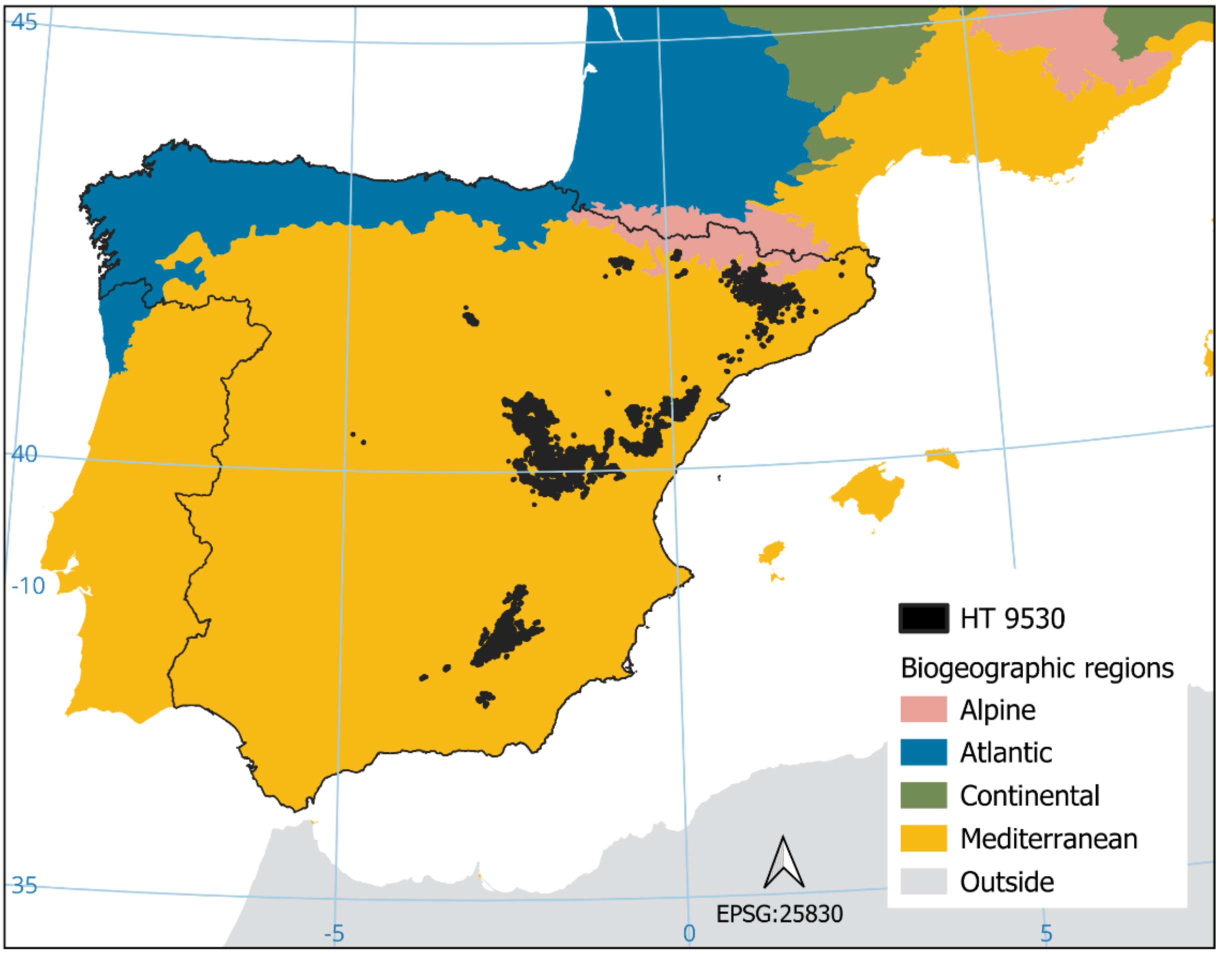
A worked example is presented below to illustrate the computational steps. The dataset, procedures and interpretations can be checked in the references given above.

Observed distribution of the habitat type

The habitat type used for this example is 9530 Mediterranean pine forests with endemic black pines, Subtype Pal. 42.63 Salzmann's pine forests. It is described in the EUNIS classification as 'Forests of the montane-Mediterranean level, on dolomitic substrate (high tolerance to magnesium), dominated by pines of the *Pinus nigra* group, often with a dense structure'.

The study area is mainland Spain. The observed distribution is based on polygons from the Spanish Forest Map at 1:50,000, with further sorting from supplementary databases to identify monospecific forests of this habitat type. These polygons represent structural forest patches defined by total and only-tree canopy density, structural vegetation type, spatial grouping of vegetation, up to three dominant species and their cover density, land use, etc. Therefore, a continuous extent of habitat type may consist of several contiguous polygons depending on respective structural variations. These polygons make the patches for this analysis.

Figure 2. Distribution of Habitat Type of Community Interest 9530 Mediterranean pine forests with endemic black pines, Subtype Pal. 42.63 Salzmann's pine forests in mainland Spain



Patches are shown at coarse scale to facilitate visualisation.

The observed distribution totals 378,111 ha and consists of 5,265 patches with a median size of 37.03 ha; first and third quartiles are respectively of 13.96 ha and 91.35 ha. The frequency distribution is extremely biased to the right, with very few large patches.

Extraction of class size frequencies

The size classes of a formed to find the pairs of values should be relevant for the logarithmic numerical scale where regression will be fitted. This discards regular size intervals of a. Instead, a simple solution is to generate a geometric progression with a common ratio of 0.5 and starting with the maximum value of a. This results in 24 pairs of values as presented in Table 1.

Table 1. Pairs of values formed by the number of patches $(n(a))$ larger than a given size (a) in m^2

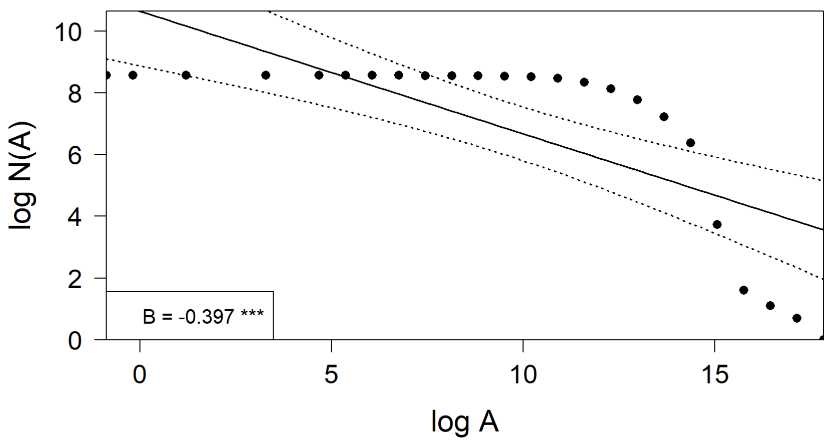
Each a value represents a size class and results from multiplying the next one by 0.5, starting by the maximum in row 24. Observe that this progression has gaps in rows 1 through 4, caused by overabundance of very small patches. Logarithmic transforms are shown on the right of the table.

ID	a	n(a)	log (a)	log (n(a))
1	0.4	5265	-0.8767	8.5688
2	0.8	5264	-0.1836	8.5686
3	3.3	5263	1.2027	8.5685
4	26.6	5262	3.2821	8.5683
5	106.5	5259	4.6684	8.5677
6	213.1	5256	5.3616	8.5671
7	426.1	5248	6.0547	8.5656
8	852.3	5237	6.7479	8.5635
9	1704.5	5222	7.4410	8.5606
10	3409.0	5210	8.1342	8.5583
11	6818.0	5183	8.8273	8.5531
12	13636.0	5136	9.5205	8.5440
13	27272.0	5033	10.2136	8.5238
14	54544.1	4775	10.9068	8.4711
15	109088.1	4230	11.5999	8.3500
16	218176.3	3427	12.2931	8.1394
17	436352.6	2395	12.9862	7.7811
18	872705.2	1370	13.6794	7.2226
19	1745410.3	587	14.3725	6.3750
20	3490820.6	42	15.0656	3.7377
21	6981641.3	5	15.7588	1.6094
22	13963282.5	3	16.4519	1.0986
23	27926565.0	2	17.1451	0.6931
24	55853130.0	1	17.8382	0.0000

Fit of Korcak's power law and residual inspection

A linear regression fitted to the 24 pairs of values (log (a), log (n(a)) is shown in Figure 3. The B Korcak exponent resulted highly significant and in a strict sense the fit could be accepted. However, the residuals are not randomly distributed along the line. There is an obvious slope change around 10 < log (A) < 15, with gentle and steep slopes on the left and right sides of this breakpoint respectively.

Figure 3. Linear regression fitted to pairs of values relating number of patches (log $N(A)$) greater than a certain size (log (A)) of habitat type 9530 in mainland Spain



Dashed lines indicate the 95% confidence interval. Slope B was significant ($p < 0.001$).

Fit of segmented regression and determination of patch size threshold

The regression in Figure 3 is clearly a candidate for a segmented fit with two segments well differentiated around an abscise value that appears to be between 10 and 15. Whilst this breakpoint might be detected through an educated guess, it is more objective, reproducible and less time consuming to perform a systematic analysis.

The overall idea is to make all possible pairs of segments from the original range of log (A), then to fit a segmented regression to each pair, and finally to select the pair with the lowest Residual Standard Error (RSE) as the best fit.

A linear regression must be fitted to at least three points. Using the ID in Table 1 to identify pairs of values, all possible pairs of segments with the available 24 pairs of values are:

1 through 3, 4 through 24

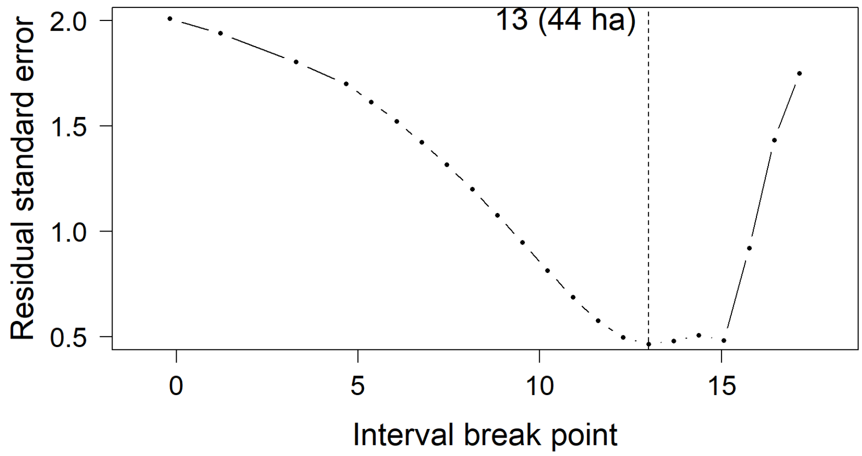
1 through 4. 5 through 24

…

1 through 21, 22 through 24

This yields 22 pairs of segments, each pair being identified by the lower log (a) of the right-hand segment, also called breakpoint. After fitting the corresponding segmented regressions, the resulting RSE values show a minimum at a breakpoint of log (a) = 12.9862 ≈ 13 (Figure 4).

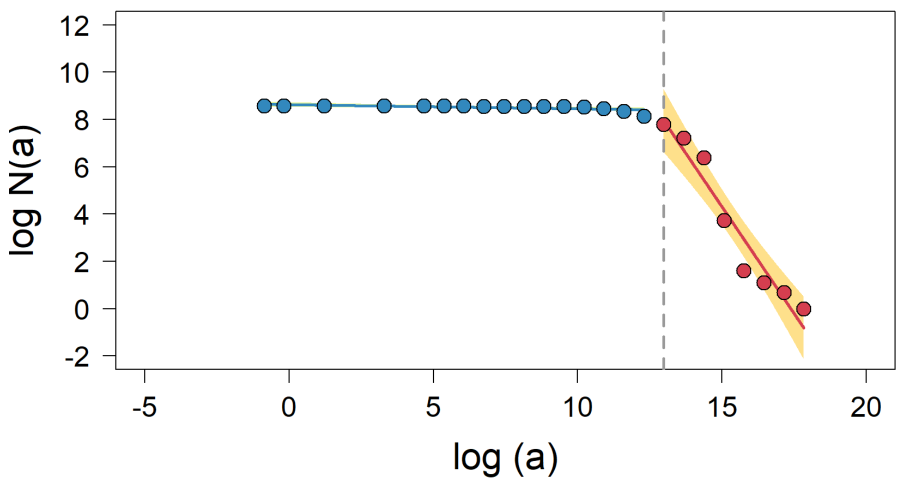
Figure 4. Residual Standard Error (RSE) of segmented regressions fitted to all possible pairs of segments made with 24 pairs of values



Each pair is represented by a dot corresponding to the lower limit of log(A) in the right-hand segment, also called breakpoint. The minimum RSE corresponds to a breakpoint of log(A) = 13, which equals to 44 ha after reverting the logarithmic transform.

Before proceeding further with any interpretation, it is necessary to determine whether the segmented regression fitted to this pair of segments is significantly better than the single regression fitted to all the range of log (A) as shown in Figure 3. This is done by an analysis of variance of the two regression models with the null hypothesis that there is no significant difference between them. In this case, this hypothesis is rejected (F = 201.3022, *d.f.* = 2, p = 5.6359E-14), and the segmented regression is accepted (Figure 5).

Figure 5. Segmented regression fitted to the log-log transformation of number of patches (log $N(a)$) greater than a certain size (log (a)) of habitat type 9530 in mainland Spain



The breakpoint separates the two segments (blue and red) around log(A) = 13.

According to Table 1, the detected breakpoint of log (a) = 12.9862 corresponds to the pair of values with ID 17, which defines segments (1 through 16, 17 through 24). Reverting the logarithmic transform of the breakpoint results in 43.6 ha, which is the patch size threshold (At). The implication is that the abundancy of large forest patches of habitat type 9530 drops abruptly for sizes greater than 43.6 ha.

Further exploiting Table 1, 2395 forest patches are equal to, or greater than 43.6 ha, and (5265 – 2395 =) 2870 patches (45.5% of the total) are smaller than this threshold. A separate query to the source database for the extent associated with this fraction of small patches resulted in 50,978.0 ha, or 13.5% of the observed distribution.

3.3.5 Decision

The patch size threshold (*At*) of 43.6 ha should now be compared to an estimate of Minimum Area Threshold (*MAT*) to decide, according to Figure 1, if an early warning of fragmentation should be issued (case: $At > MAT$) or if definite habitat fragmentation should be accepted and the corresponding 13.5% of the observed distribution should be detracted from the stock of habitat assets (case: $At \leq MAT$).

Unfortunately, it can be advanced that a proper *MAT* estimate is not currently available. However, it is worth exploring some sources, if only to raise awareness of the urgent need to enhance the empirical measures available for Annex I habitats.

The first and most obvious resource for the required information should be the web tool on biogeographical assessments of conservation status of species and habitats under Article 17 of the Habitats Directive (https://nature-art17.eionet.europa.eu/article17/). For habitat 9530, the data reported by Spain gives a surface area covered of 2,944 ha in the Alpine region and 404,312 ha in the Mediterranean region. The discrepancy between the combined surface of 407,256 ha from this source and the extent of 378,111 ha used in this exercise can be attributed to the earlier date of the status report (2013-2018) with respect to the CHFE50 maps used here, which were commissioned in 2019 (Sánchez de Dios et al., 2019).

In both cases, the Favourable Reference Area (FRA) is annotated as ‘more’ than the reported current surfaces. This is not helpful, both due to the lack of specificity and the unexplained reasons that may support this value judgment. Even if FRA were given as a proportion of the observed distribution, as mentioned in Section 3.3.3 for Habitat Type 9360, differences in scale would make this information of a limited value, but at least a hint on relationships between biodiversity and area would be given.

In view of these limitations, the best option is to fit species to area curves from field data at a spatial scale that is commensurable with patch sizes in the observed distribution. This is not necessarily as onerous as it seems. In Spain there is the fourth update of the National Forest Inventory (IFN, for *Inventario Forestal Nacional*)[1]. It consists of small field plots (10 m radius) stratified by provinces (NUTS-3) and then by forest types and masses. It covers all the country, and the plots are fixed, therefore they are revisited every ten years. Tree and shrub species in each plot are recorded in addition to many other forest and dendrometry variables. Therefore, the information is there in this case to build the required species to area curves by aggregating plots. Such curves will of course be valid only for ecosystem components linked to the spatial scale of the described sampling, mostly plants. This is fine for the objectives of vegetation

[1] Available at: https://www.miteco.gob.es/es/biodiversidad/temas/inventarios-nacionales/inventario-forestal-nacional.html

cover fragmentation. However, insects or mammals, just to name two examples, are likely to require dedicated sampling designs.

An even more promising approach to use the IFN to assess habitat condition was developed by Pescador et al. (2022). They used a set of variables from the IFN with a diagnostic value on conservation status for the Structure and Function parameter, and selected IFN plots that could be allocated to forest Annex I habitats to yield full conservation assessments by habitat type. Whilst this study focused on whole habitat types, an equivalent methodology could be applied to individual patches. It could be hypothesized that the larger the patch size of a certain forest habitat type, the better its conservation status will be in terms of those variables. If this relation proves true, it could be used to estimate MAT from an explicit relationship between patch size and conservation status.

Further development of these two approaches based on the Spanish IFN is beyond the scope of this document. They are described here only to provide objective and explicit avenues to get MAT, a parameter that is crucial to assess risks associated with fragmentation and useful for many other land management objectives. Equivalent forest inventories might exist in other member states that, either alone or with little additional effort, may be used for this purpose. Further research on the potential of existing data bases for fragmentation studies, as well as cost-effective enhancements to current sampling schemes, are warranted.

4. Next steps to address future needs

The greatest goal of methodological approaches to fragmentation of habitat types in the EU is, probably, to make a concrete and quantitative contribution to the Evaluation Matrix for assessing conservation status of a habitat type (Article 17 of the Habitats Directive). Such a contribution should be neat and have the potential to shift, if necessary, the conservation status beyond subjective expert judgements.

The Evaluation Matrix does not have a component for fragmentation specifically, which is appropriate because the relevance of fragmentation is in its impacts on the parameters of the matrix. The example developed in the Section 3.3.2 shows a plausible mechanism by which fragmentation may have an impact on the parameter Area, and the hint given in Section 3.3.5 suggests how applying assessments of habitat condition at the patch level could impact two parameters, Area, and Structure and Function.

Member states reporting under Article 17 should incorporate feedback mechanisms articulated under some conservation accountancy framework, preferably harmonized with the UN System of Environmental-Economic Accounting. This would have the important implication that quantities and parameters from the six yearly reports would be comparable, directly controlling the central concepts of habitat assets and habitat stocks. This way, the reporting under Article 17 would become proactive with a return period of six years, which reflects the fast changes affecting European habitats.

In the short term, there are certain steps that should be taken:

- A catalogue of methods dealing with fragmentation assessment and with a potential to be used across EU-27 should be made. The concept of Minimum Area Threshold is central to these methodologies, and should receive particular attention, especially for its multiscale implications. Specifications on input data, metrics, software and intended application of the results should be given, so that the catalogue itself becomes a first step towards standardization. Its declared objective should be that those methods could be available to all member states to complement the approaches they are currently applying.
- A catalogue of data with potential to be used for fragmentation assessment should be made closely linked to the methods catalogue. It is urgent to know if, and in which terms, certain critical datasets are available at the member state level. This includes observed and predicted habitat distributions, field sampling schemes of certain habitat types, etc. In the case of forests, mapping guidelines should specify minimum stand age, minimum patch area discontinuity criteria, etc. It is noteworthy that many of those datasets are critical for many ecological studies beyond fragmentation. Data interoperability should receive special attention.
- If these two catalogues were available, it would make an EU-wide assessment of fragmentation at the level of habitat type possible. It is important to remark that there is already an assessment at the landscape level (European Environment Agency, 2011), but because it is at the landscape level it has limited use for the Evaluation Matrix of specific habitats. The proposed fragmentation assessment would have, among other objectives, to clarify spatial scales and organization levels at which fragmentation results can be aggregated. The conventional progression from NUTS levels to EU to biogeographic region works well for ecological attributes that can be lumped (e.g., net primary productivity, biomass, surface area occupied by a habitat). However,

aggregation is the opposite of fragmentation, and how to present results for increasingly larger extents is not obvious.

- A transversal panel of experts guiding the development of these recommendations and providing training to managers and technicians at member state level would be necessary. Probably a flexible group made by a few experts and where member states may adhere voluntarily is a more realistic proposal than creating a comprehensive structure involving all member states. It is important to acknowledge that the points mentioned here should be achieved through evolution rather than by design.

Habitat fragmentation is likely the greatest challenge faced by natural and seminatural landscapes of Europe. Its impact on ecosystem functions is indirect in the sense that it affects persistence and resilience rather than performance. This may be a reason why it is often tackled without a crisp causal rationale or even overlooked altogether. Notwithstanding this, fragmentation is as dynamic as the society creating it, and this means a fast pace in the European space. It is urgent to incorporate its understanding and measurement into a management decision framework.

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