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VEGETATION SCIENCE AND HABITATS DIRECTIVE

A bottom–up approach for the conservation status assessment of structure and functions of habitat types

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Abstract

Monitoring of habitat types conservation status is an essential task in the frame of the European policy for biodiversity conservation. The parameters to be assessed for the purposes of habitat types' conservation status assessment are described in several European documents, but the methodology for their determination has not yet been standardized or optimized. This study presents methods for the assessment of the actual status and the future prospects of structure and functions of habitat types. Specifically, it presents a bottom–up approach for the assessment of these two parameters at different spatial scales. In the proposed method, conservation status assessment is based on a classification of habitat types to subtypes, with the latter representing the basic monitoring entities. The conservation status is assessed by recording: (i) the presence/absence of specific indicators of structure and functions per habitat type, and (ii) the presence/absence, abundance, and vitality of the typical species of the habitat subtypes. The typical species are determined objectively using algorithms and fidelity coefficient values. The conservation status and future prospects of structure and functions (including the typical species) are estimated quantitatively with the help of numerical methods and algorithms, but their assignment to conservation status classes is based on thresholds defined by experts. Assessments are made at the local scale, but can be upscaled to coarser ones (up to the national level). The proposed methods have been applied in Greece and were effective both in terms of results obtained and costs needed.

Keywords Biodiversity · Greece · Habitats Directive · Indicator species · Monitoring · Vegetation

1 Introduction

All EU Member States (MS) should monitor and assess the conservation status of habitat types of community interest according to the 92/43/EEC Directive (Habitats Directive); and for this reason, a large number of specialists are occupied in various monitoring projects at regional level to national level (Lengyel et al. 2008a). We assume that readers are familiar with the terms and procedures of the conservation status assessment, as they are set by the Habitats Directive (92/43/EEC) and are presented in European Commission (2011) and in Evans and Arvela (2011); for this reason, we present here only the basic principles. The assessment of the conservation status of habitat types is based on the evaluation of three parameters, namely 'Structure and Functions', 'Area', and 'Range'. The parameter 'Structure and Functions' includes the assessment of 'Typical Species'. Each of these parameters is assessed regarding its current and future status and its future trend. From the combination of the current and future status, as well as of future trend, a fourth parameter

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is developed, namely ‘Future Prospects’. Every MS has to quantify these four parameters to assess the conservation status of habitat types. Approaches and examples for the assessment of each parameter are presented in Evans and Arvela (2011) and in European Commission (2011).

These general principles allow for a great variation of methodologies that can be applied. In addition, indeed, different MS quantified these parameters using different approaches. This variation of monitoring methodologies throughout Europe has been already identified a decade ago (Cantarello and Newton 2008; Lengyel et al. 2008a; Schmeller 2008); yet, still today, objective (e.g., numeric and quantitative) methods to assess each parameter are lacking, allowing for different interpretations. Differences arise at all stages of monitoring implementation: from sampling design to data analysis. The issue of comparability of the monitoring methods is also mentioned in the report of European Environment Agency (2015), about the results from reporting under the nature directives for the period 2007–2012, as a cause of bias that complicates the comparison of conservation status assessment between MS, as well as among different reporting periods.

The reliability of any monitoring project is largely based on the standardization of methods and the development of a ‘standard operating procedure’ (Hill et al. 2005). This standardization should accurately describe each step of the monitoring procedure, from data collection to data analysis and reporting, to reduce subjectivity and discrepancy among observers and among years during the implementation of a monitoring project (Hill et al. 2005). Furthermore, another important issue in the monitoring projects is the implementation of quantitative methods and the avoidance of assessments by expert judgment (Carignan and Villard 2002; Kovač et al. 2016; Yoccoz et al. 2001). The application of quantitative methods is considered as crucial in all the steps of a ‘systematic conservation planning’ (e.g., the measuring of biodiversity surrogates, the setting of conservation targets), since this is the only way to deal with the uncertainty which is involved in all the steps of conservation planning and to improve conservation planning through an optimization procedure (Margules and Pressey 2000).

To standardize monitoring methods, the definition of monitoring parameters should first be standardized. Although the Habitats Directive was adopted in 1992, a clear definition regarding the parameter structure and functions as well as of typical species is still missing (Evans and Arvela 2011). Structure includes all the physical components of a habitat type formed by species (both living and dead) and functions concern the ecological processes occurring at a number of temporal and spatial scales within a habitat type (Evans and Arvela 2011). This parameter should include characteristics of the habitat type that indicate healthy ecosystem (e.g., diversity of dominant species age classes and rich understory plant diversity) or lack of indications of anthropogenic degradation (e.g., no apparent signs of logging or planted species). There is a plethora of publications describing with more or less detail some structural characteristics of habitat types (Carli et al. 2016; Davis et al. 2014; Del Vecchio et al. 2016; Hill et al. 2005; Joint Nature Conservation Committee 2013; Kovač et al. 2016; Søgaard et al. 2007), yet this parameter still remains abstract.

The second component of structure and functions in Habitats Directive is the typical species. Likewise, there is no clear definition of typical species neither in the Habitats Directive nor in the explanatory notes and guidelines that have been published for the assessment and the reporting under article 17 of the Directive (European Commission 2006; Evans and Arvela 2011). Typical species may include all species groups; for example, vascular plants, lichens, bryophytes, as well as all animal groups. Different MS used different approaches to define typical species. For example, in France, typical species were considered as indicative species of ecosystem’s functions, thus focusing on species functional traits (Maciejewski 2010). However, in European Commission (2006), typical species are related to indicator species according to the phytosociological approach and thus to the characteristic and/or the differential species of the associations as well as of the higher level syntaxa (alliances, orders, and classes) (e.g., Braun-Blanquet 1964; Dierschke 1994; Leuschner and Ellenberg 2017a, b). According to both versions of the explanatory notes and guidelines (European Commission 2011; Evans and Arvela 2011) and other references (e.g., Carignan and Villard 2002), typical species should reflect the favourable structure and functions of the habitat type and should be sensitive to changes to comprise early warning indicators. Furthermore, according to Carignan and Villard (2002), the most important characteristics that indicators may possess are: (i) to provide early warning for changes, (ii) to indicate the cause of changes rather than just the existence, (iii) to represent the full gradient of possible changes, and (iv) to be cost-effective and measurable even from non-specialists.

The conservation status assessment is not based only on structural and functional characteristics (including typical species) of a habitat type. Parameters regarding the area and the range of a habitat type are equally important. These are mainly assessed at the regional to national scale applying methods of vegetation mapping and habitat modeling, including the use of remote sensing (Buchanan et al. 2008; Nagendra et al. 2013; Spanhove et al. 2012; Vanden Borre et al. 2011). However, structure and functions, typical species, and the identification of the occurrence of pressures and threats are better evaluated at the local scale (Søgaard et al. 2007). Therefore, it is essential to upscale and aggregate the outputs obtained at the local-scale assessment to combine them with other parameters already assessed at broader spatial scale.

During 2014–2015, in the framework of a national monitoring project of habitat types, a great effort was invested in Greece to develop and apply a harmonized procedure for the standardization of the conservation status assessment methods. Specifically, standardized numerical methods have been developed and tested for the assessment of structure and functions (including typical species) of habitat types and their future prospects. In addition, a bottom–up methodology was developed: (i) to take advantage of the synchronous in situ data collection over the entire NATURA 2000 (N2000) network in Greece for the estimation of the conservation degree at the local scale, and (ii) to aggregate the data to assess the conservation status of the habitat types at the national level. The term conservation degree is used hereafter for the conservation status assessment at the local scale (e.g., sampling locality) or regional (e.g., N2000 sites) scale, while, for the national scale, the term conservation status is applied. This need for differentiation in the terminology has been proposed by Evans and Arvela (2011) as well as by Chrysopolitou et al. (2015) for Greece to distinguish the assessments made in local/regional scale for the completion and/or update of Standard Data Forms, from those made at the national scale or biogeographical scale for the purposes of monitoring according to Article 17 of the Directive 92/43/EEC. The methods described in this study cover the entire assessment procedure, from the preparation of data collection, up to the analysis for the final assessment of the conservation status, including the methods applied to transfer the assessment of parameters through different spatial scales.

The aims of this paper are: (i) to present the methods implemented in Greece for the conservation status assessment of structure and functions of habitat types, (ii) to present the approach used for the aggregation of the assessment from the local to the national scale, and (iii) to discuss the rationale of the methods selection, their characteristics and benefits, but also their pitfalls and aspects that need to be improved.

2 Data collection planning

The adoption of a bottom–up scheme for the assessment of the conservation status allows for and requires the collection of data at the local scale usually by different researchers. However, even when the same methods are applied, various vegetation properties can be estimated with high uncertainty when different researchers collect data (Archaux 2009; Bergstedt et al. 2009). It is safe to assume that bias, uncertainty, and subjectivity in the conservation status assessment increase with the absence of a standardized protocol designed for the collection of the appropriate data. To put it in other words, properly designed field data sheets assure consistency and decrease time needed for in situ observations (Hill et al. 2005). The design of the field data sheets should be compatible with the database structure and should allow for the collection of all required data to fulfil the needs for the habitat types monitoring and reporting (Article 17, Dir. 92/43/EEC).

2.1 Determination of typical species of habitat types

The diagnostic species of the vegetation units (syntaxa) included in each habitat type are considered as typical species of the habitat type. We determined de novo the typical species of the habitat types occurring in Greece applying a series of analyses, as shown in Fig. 1 and described below.

We classified approximately 18,000 vegetation plots from all over Greece, representing all the habitat types recorded so far in the country, by means of cluster analysis using the Ward's linkage method (Ward 1963) and the Lennon et al. (2001) index as distance measure. The latter index was introduced as an index of beta diversity to measure species turnover among plots but taking into account the differences in species richness among plots (Koleff et al. 2003). The use of this index was preferred after comparing the classification results obtained with other distance measures (e.g., Bray–Curtis, Euclidean). The better performance of the Lennon et al. (2001) index may be attributed to the fact that the database includes plots of different sampling effort. After inspecting the classification results, 229 vegetation groups representing 85 habitat types of Community or national interest were distinguished. Most of the habitat types were represented by more than one cluster and two of them were represented by more than ten clusters.

After the classification, we determined the differential taxa of each cluster by using the algorithm introduced by Tsiripidis et al. (2009). The advantage of this algorithm is that it performs multiple comparisons between all possible combinations of clusters in a data set, and consequently, it determines the diagnostic species of syntaxa of different hierarchical levels. For the differential species, we calculated the phi coefficient using the percentage frequency of taxa in the distinguished clusters. The phi coefficient was calculated for the clusters differentiated positively against the ones that were differentiated negatively or not being differentiated. As the calculation of the phi coefficient by the above-mentioned way results in one value for all the clusters differentiated positively, we relativized this value by multiplying it by the ratio of the frequency of each taxon in each cluster to its maximum frequency in the table of clusters. Finally, we also calculated the phi coefficient following the way proposed by Chytrý et al. (2002), using the percentage frequencies of taxa in the clusters.

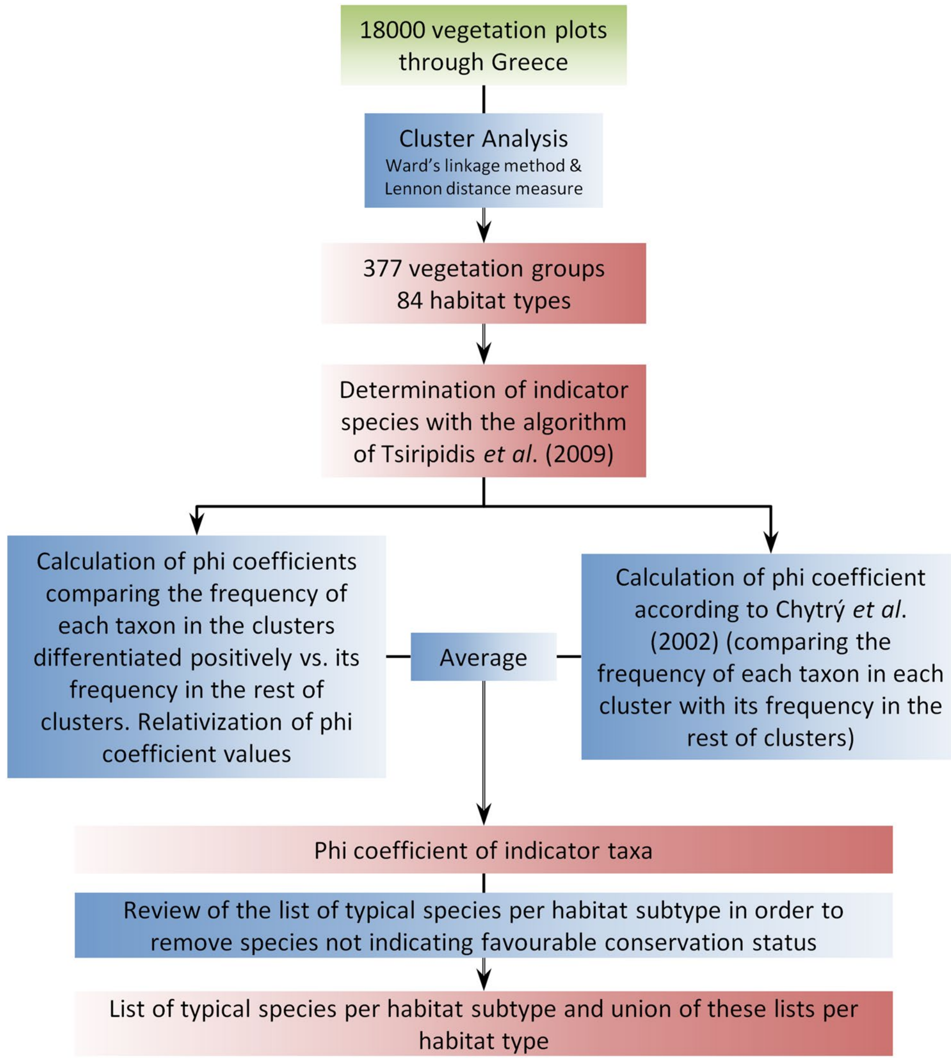


Fig. 1 Steps to determine typical species for the habitat types occurring in Greece. Background colour legend: light green fading towards down indicates data, blue fading towards right indicates analyses or procedures, and light red fading towards left indicates outputs (colour figure online)

The latter compares the frequency of each taxon in each cluster with the corresponding frequency in all the other clusters. This gives more weight to the 'absolute differential species', i.e., the ones occurring almost exclusively in one or at most in very few clusters. To take into account both types of differential species (those differentiating a large number of clusters and those differentiating one or few clusters), we calculated the average value of phi coefficient calculated using the former two methods, for all the taxa found as differential by the algorithm of Tsiripidis et al. (2009).

Thus, a list of typical species was created for each cluster. Experts in each vegetation type reviewed this list to exclude taxa which exhibit preferential occurrence in certain vegetation types, but they cannot be used as indicators of favourable conservation status. An example of such a species is *Amorpha fruticosa*, which, although occurring only in riparian habitat types, it is an alien species often indicating intense disturbance in these ecosystems.

As an example, the typical species of the three subtypes determined for the priority habitat type 91E0 [Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (Alno-Pandion, Alnion incanae, Salicion albae)] are shown in Online Resource 1.

As all the vegetation plots in the database included also information about the sampling locality, the final lists of typical species correspond not only to certain plant communities but also to certain geographical areas (see Online Resource 2).

Therefore, the field evaluators used a specific predefined list of typical species depending on the habitat type and the geographical area which they sampled. However, the field researchers had to verify that the suggested list of typical species was appropriate for each sampling locality, especially in cases where more than two lists of typical species were obtained for the same habitat type in adjacent areas. Furthermore, in cases of doubt, the field researcher was advised to check the presence–absence of taxa using a general typical species list which was the union of all the species lists of the subtypes of the habitat type.

2.2 Specific structure and functions

Unlike typical species, we consider that structure and functions have a more 'global' character and that the same list of indicators of structure and functions can be applied for a habitat type throughout its distribution within the country. Furthermore, some indicators of structure and functions can be shared among different habitat types.

We developed lists of specific structural characteristics and functions for each habitat type that can serve as indicators of favourable conservation status taking into account relevant publications where similar structural and/or functional indicators have been proposed (Cantarello and Newton 2008; Carli et al. 2016; Davis et al. 2014; Del Vecchio et al. 2016; Hill et al. 2005; Joint Nature Conservation Committee 2013; Kovač et al. 2016; Søgaard et al. 2007). An example of such a list for the habitat type 91E0 is given in the Online Resource 3. Field evaluators in each sampling locality were asked to check which of these indicators of structure and functions are present or not.

2.3 Pressures and threats

Regarding pressures and threats, we adopted the methods proposed in (Evans and Arvela 2011), taking into consideration the list of pressures and threats available in the Web page of the European Topic Centre on Biological Diversity (http://bd.eionet.europa.eu/activities/Natura_2000/reference_portal, assessed on March 2014). Pressures are past or present ongoing impacts that threaten the long-term viability of a habitat type, while threats are similar impacts but refer to the foreseeable future. The importance of pressures and threats identified in the field was graded using the three-grade ordinal scale ('high', 'medium' and 'low'), depending on their intensity and extent of occurrence. Furthermore, if management measures (e.g., restoration of natural flooding regime in riparian areas) have been applied in an area to deal with specific pressures or threats, they were recorded as positive impacts on the conservation degree of a habitat type. The positive impacts were considered if they can counterbalance the negative effect of certain existing pressures or threats. Pressures and threats as well as positive impacts were used for the estimation of the future trend of the habitat type's conservation status and, consecutively, for the estimation (on the basis of expert judgement) of the habitat type's future conservation degree.

2.4 Field sheets

Different field sheets were prepared for each habitat (sub) type. The field sheets included: (i) the habitat (sub)type's list of typical species, (ii) the habitat type's list of indicators of structure and functions, and (iii) fields to record the observed pressures and threats alongside their intensity, as well as any positive impact. In addition, in the field sheets, evaluators recorded information regarding the sampling locality. This information included: (i) position of sampling locality (e.g., geographical coordinates, name of locality), (ii) general ecological and structural features (total cover of each vegetation layer, elevation, exposition, inclination, and soil properties), (iii) data regarding the evaluator identity, and iv) general remarks (e.g., occurrence of invasive species, disturbances, and adjacent vegetation types). Finally, each sampling locality was accompanied by at least two photographs depicting aspects of the sampled area. An example of such field sheet for the subtype C of habitat type 91E0 is given in the Online Resource 4.

3 Assessment of the conservation degree

Having collected the data with a standardized approach, next step includes the application of methods for the objective assessment of each parameter at the local scale. Namely, hereafter, we present the methods adopted for the evaluation of the actual conservation degree of structure and functions (including typical species) and the future prospects of structure and functions.

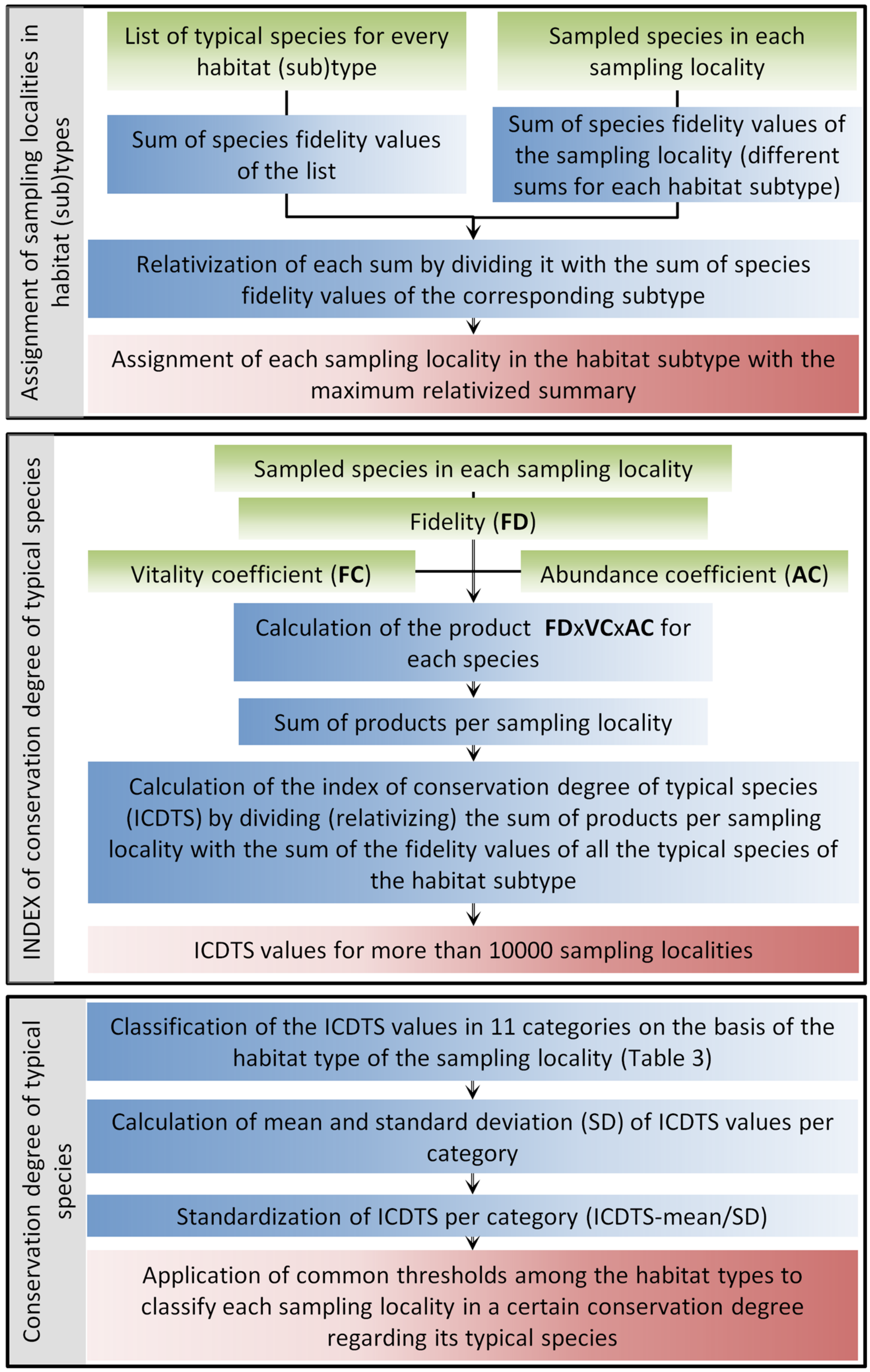
3.1 Assessment of actual conservation degree of typical species

The analysis of sampled data for the assessment of the conservation degree of typical species consisted of three distinct steps, each including a series of analyses, as shown in Fig. 2.

3.1.1 Assignment of sampling localities to habitat subtypes

The assessment of the conservation degree of typical species is based on the comparison of the species recorded at each sampling locality with the list of typical species of a certain habitat subtype. Therefore, it is essential to check if all sampling localities were classified in the correct subtype (this was necessary especially for the cases that the field researchers used the general list of typical species for a habitat type). For this purpose, we summed the fidelity values of

Fig. 2 Actual conservation degree assessment of typical species. Background colours indicate the same as in Fig. 1 (colour figure online)



the typical species recorded in each sampling locality. The sums calculated for each sampling locality were as many as the subtypes of the habitat type in which the sampling locality was assigned. For each of these sums, only the common taxa between those recorded in the sampling locality and each subtype were taken into account with the fidelity values that the taxa present in the specific subtype. The use of fidelity in the calculation of the sums aimed to take into account the ecological and geographical affinity of species with the habitat subtypes. The sum for each habitat subtype was then divided by the sum of fidelity values of all typical species of the subtype. This type of relativization was applied to make the sums of species fidelity values independent from the number of typical species per habitat subtype. Each sampling locality was eventually assigned to that subtype for which it presented the maximum relativized sum. Actually, the above calculations correspond to a Jaccard similarity coefficient weighted by the species fidelity values for the habitat subtypes. However, the species recorded in the sampling locality but not included in the typical species of the subtype are actually omitted from the denominator of the formula of Jaccard index, because they have a zero fidelity value in the habitat subtype. An example of the above-mentioned calculations is given in the Online Resource 5.

3.1.2 Estimation of index of actual conservation degree of typical species

For the assessment of the actual conservation degree of the typical species in each sampling locality, the fidelity value of each taxon was multiplied by coefficients related to its abundance and vitality. Thus, field evaluators were advised to record (i) the abundance of the typical species found in each sampling locality using the AFOR scale (A = Abundant, F = Frequent, O = Occasional, and R = Rare), and (ii) their vitality (Braun-Blanquet 1964; Dierschke 1994) using three categories [1: feeble plants that cannot reproduce (either by seeds/spores or vegetatively) and be developed in all the phenological (e.g., develop flowers or seeds) or growing stages (e.g., develop normally in height) in comparison with their biology, 2: plants with a poor reproduction or development of certain phenological or growing stages in comparison with their biology, and 3: well developed and reproduced plants in comparison with their biology].

The abundance coefficient was calculated by comparing the abundance of each taxon occurring in each sampling locality with its average abundance in the subtype, as it was calculated on the basis of the plots of the database classified in the subtype. The average abundance of each taxon in the subtype was transformed in the scale AFOR and the values 4, 3, 2, and 1 were given for the four grades (A, F, O, and R) of the scale. If the abundance of the taxon in the sampling locality differed for more than one grade from that in the subtype, the coefficient was equal to the ratio of the abundance in the sampling locality to that in the subtype. For example, if the abundance of the taxon in the sampling locality was 1 and its average abundance in the subtype was 3, then the coefficient was equal to 1/3. If there was no difference in the abundance or it was up to one grade of the AFOR scale, the coefficient was set to 1.

The vitality coefficient value was set equal to 1, 0.66, and 0.33 for the vitality categories 3, 2, and 1, respectively.

For each sampling locality, we summed the products of the species fidelity values with the abundance and vitality coefficients values and the sum was relativized by dividing it with the sum of fidelity values of all the typical species of the habitat subtype to which the sampling locality was classified. This value was considered to represent the conservation degree of the typical species in each sampling locality and it is named hereafter as the index of conservation degree of typical species (ICDTS).

3.1.3 Assessment of the actual conservation degree of typical species

The values of ICDTS were calculated for all sampling localities, which were more than 10,000. Habitat types were then divided in 11 categories (Table 1) which correspond to a great extent to the categories identified in the Habitats Directive. Within each of these 11 categories, the average and standard deviation of the ICDTS values were calculated. For each sampling locality, we calculated the difference of its

Table 1 Habitat types grouped in categories on the basis of their physiognomy and ecology. The average and the standard deviation (SD) of the index of conservation degree of typical species are presented for each category

Category	Average of ICDTS	SD of ICDTS	Habitat type codes
1	0.291	0.143	1210, 1240, 1310, 2110, 2120, 2190
2	0.275	0.138	1410, 1420, 1430, 1510, 2220, 2230, 2250, 2260
3	0.232	0.205	3130, 3140, 3150, 3170, 3240, 3260, 3280, 3290
4	0.204	0.112	4060, 4090
5	0.295	0.144	5110, 5210, 5330, 5420, 5430
6	0.207	0.151	6170, 6220, 6230, 62A0, 62D0, 6420, 6430, 6510
7	0.263	0.136	7140, 7210, 7230
8	0.211	0.182	8140, 8210, 8220, 8320
9	0.325	0.131	91E0, 91F0, 92A0, 92C0, 92D0
10	0.374	0.164	9110, 9130, 9140, 9150, 9180, 91CA, 91M0, 9250, 9260, 9270, 9280, 9410, 9530, 9560, 95A0
11	0.536	0.238	2270, 9290, 9310, 9320, 9340, 9350, 9540

Table 2 Key to assessment of conservation degree of typical species at the sampling locality level

Conservation degree	Value of ICDTS
Favourable (FV)	Plot's ICDTS > average − 1 × SD
Inadequate (U1)	Average − 1 × SD ≥ Plot's ICDTS > average − 1.5 × SD
Bad (U2)	Plot's ICDTS ≤ average − 1.5 × SD

Table 3 Key to assessment of conservation degree of structure and functions at the sampling locality level

Conservation degree	Proportion of indicators marked as present
Favourable (FV)	≥ 50%
Inadequate (U1)	< 50% but ≥ 25%
Bad (U2)	< 25%

ICDTS value from the average of the category of the habitat type which it belongs. This difference was then divided by the standard deviation (SD) of the ICDTS of the habitat type category. In this way, we calculated how many SDs each sampling locality's ICDTS differs from the average of the category of the habitat type which it belongs. These values are comparable between the sampling localities regardless the habitat type which they belong. The distinction of different categories of habitat types and the calculation of averages and SDs per category was applied, because the range of ICDTS index values differs according to the habitat type ecology and the number of typical species. Ideally, the average and SD of ICDTS values should be calculated for each habitat type, but, in our data set, there were some habitat types with a relatively low number of sampling localities. Furthermore, increasing the number of sampling localities used to calculate the above-mentioned statistics reduces the effects of outliers.

Finally, after testing with different threshold values, the ones presented in Table 2 were chosen to classify the three classes of conservation degree (FV, U1, and U2) related to the typical species. The specific thresholds were chosen, after inspecting the percentage of sampling localities which were classified in each of the three classes of conservation degree, as well as comparing the classification in conservation degrees that the thresholds produce, with the empirical estimations of the conservation degree of sampling localities during the field sampling.

3.2 Assessment of the actual conservation degree for specific structure and functions

The conservation degree for specific structure and functions was assessed on the basis of the proportion of indicators present in each sampling locality from the total number of indicators for each habitat type and the thresholds chosen to be applied are presented in Table 3.

3.3 Assessment of actual conservation degree of structure and functions (including typical species)

The actual conservation degree of structure and functions (including typical species) was assessed after combining the assessment of its two sub-parameters: specific structure and functions and typical species (Table 4).

The actual status of structure and functions (including typical species) is a parameter appearing in the habitat's report. Therefore, the assessment of the conservation degree at the sampling locality level should be upscaled to the national level using the methodology described in the respective parts of this paper.

3.4 Estimation of future prospects of structure and functions

The future trend of conservation degree of structure and functions was estimated using the number and importance of the pressures and threats recorded in each sampling locality. The trend is considered as favourable (FV), when up to one pressure/threat of medium importance and no pressure/threat of high importance is recorded or positive impacts (e.g., management measures) balance higher number or importance of pressures/threats. If there is more than one and up to three pressures/threats of medium importance, and none of high importance or positive impacts balance higher number or importance of pressures/threats, the trend is considered as unfavourable-inadequate (U1). If there are more than three pressures/threats of medium importance or at least one pressure/threat of high importance, the trend is assessed as unfavourable-bad (U2). The future conservation degree was estimated by expert judgment at each sampling locality taking into consideration the number and importance of pressures and threats, the actual conservation degree, as well as any applied or planned conservation measures in the area.

The future prospects of the conservation degree of structure and functions are then estimated on the basis of the actual degree and the future trend for every sampling locality, following the suggestions described in Fig. 6 and Table 2, pages 33 and 34, respectively, in Evans and Arvela (2011).

Table 4 Key to assessment of conservation degree of structure and functions (including typical species) at the sampling locality level

Conservation degree	Combinations of conservation degree of sub-parameters
Favourable (FV)	Typical species and structure and functions FV
Inadequate (U1)	Typical species and/or structure and functions U1, but none U2
Bad (U2)	Typical species and/or structure and functions U2
Unknown	Typical species and/or structure and functions Unknown, but none U2

4 From the conservation degree to the conservation status assessment

At the spatial scale of sampling locality, we have estimated: (i) the actual conservation degree of structure and functions (including typical species), and (ii) the local prospects of the conservation degree of structure and functions. To assess the conservation status of each habitat type at the national level, two procedures are required: (i) the upscaling of these parameters at the national level and (ii) the combination of these parameters with the ones related with the area and the range that have been assessed at the regional to national scale. The latter is already described by Evans and Arvela (2011), and thus, we will not discuss it in this paper, yet the former requires the development of a specific methodology. A broad schematic representation of the transition between spatial scales and the combination of parameters in each step is shown in Fig. 3.

Our proposed method leads to the evaluation of the conservation degree at a specific locality and, in turn, one or more sampling localities are included in a single unit of broader geographical scale (i.e., European Environment Agency (E.E.A.) reference grid cells, N2000 site, biogeographical region, MS).

The transition among spatial scales from the local (sampling locality) to the national could include one or more intermediate steps, as shown in Fig. 4. In Greece, the assessment of the conservation status was evaluated using the intermediate step of assessing the conservation degree of each habitat type at the E.E.A. 10 km reference grid cell. The field evaluators were instructed to sample at least one locality, if possible, in every E.E.A. 10 km grid cell that a habitat type was known to occur within each N2000 site.

To estimate the actual conservation degree of each habitat type at each E.E.A. 10 km grid cell, we applied the rules presented in Table 5 to the data from all sampling localities within each grid cell.

The next step includes the aggregation of outputs from the E.E.A. cell to national scale. This aggregation followed the 'majority' rule. Specifically, the actual conservation status of a habitat type at the national level was set to the same class with that of the conservation degree found in most E.E.A. cells in which the habitat type's conservation degree was assessed. In the case of tie (lack of majority), the worst conservation degree was applied.

The final choice of rule sets for the aggregation of the assessment outputs at broader scales was based on expert opinion. Alternative assessments of the conservation status using different combinations of rules for the intermediate steps were performed and the outputs were subjected to the opinion of a series of experts to define the combination of methods that best depicts the condition of habitat types in Greece. It was concluded that a strict '75-25' rule for the aggregation at the E.E.A. grid cell followed by the 'majority' rule was the best combination for the final assessment of the conservation status at the national scale.

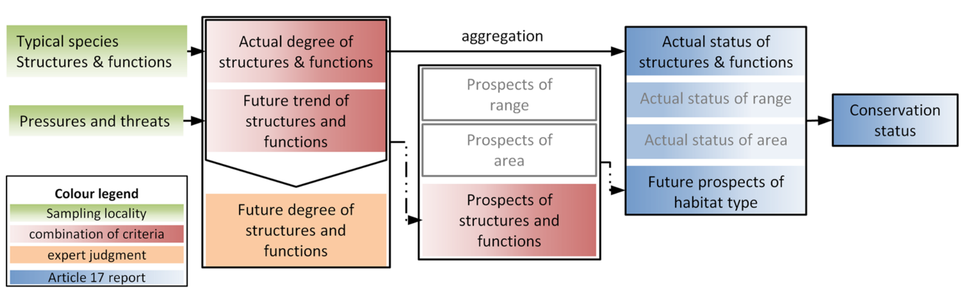


Fig. 3 Conservation status assessment parameters and transition among spatial scales. Data collected in sampling localities (light green background fading towards down) can be combined at the same spatial scale to result in the assessment of certain parameters (red background fading towards left) and further combined with parameters derived from expert judgment (orange uniform background) before they are upscaled and combined again with parameters of area and range. Data on blue background fading towards right are reported (article 17 of Habitats Directive) at the national scale (colour figure online)

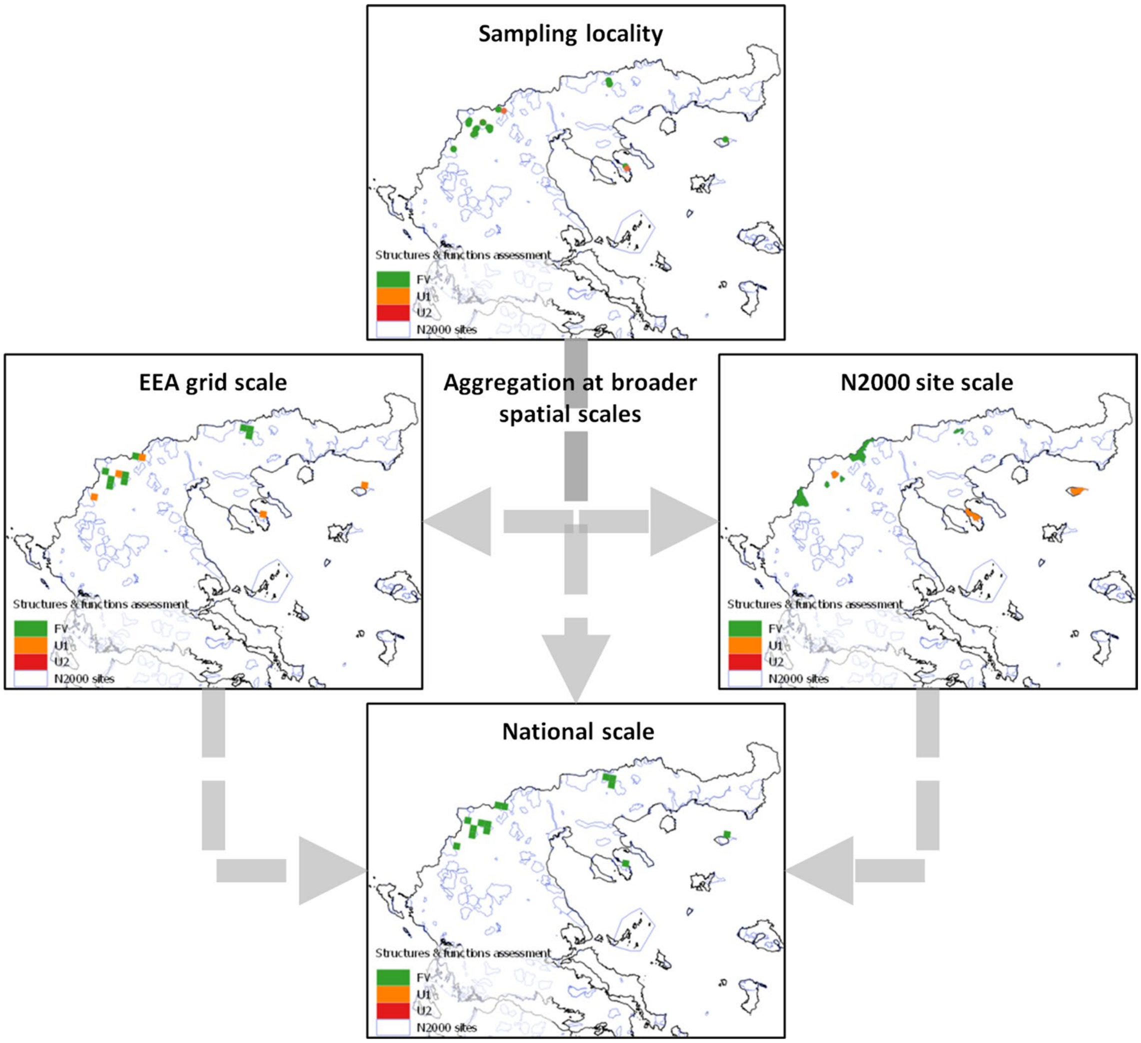


Fig. 4 Bottom–up approach allows for the consideration of (sub)parameters evaluated at different spatial scales and the aggregation of (sub) parameters assessment at broader scales, using objective algorithms (colour figure online)

Table 5 Key for the upscale of the conservation degree from the local to the E.E.A. grid cell level

Conservation degree at E.E.A. 10 km grid cell	Rule
Favourable (FV)	≥ 75% of sampling localities included in the E.E.A. grid cell evaluated as FV in actual conservation degree of structure and functions
Inadequate (U1)	Any other combination
Bad (U2)	> 25% of sampling localities included in the E.E.A. grid cell evaluated as U2 in actual conservation degree of structure and functions
Unknown	More than 25% of sampling localities included in the E.E.A. grid cell with unknown actual conservation degree of structure and functions but less than 25% of sampling localities evaluated as U2 in actual conservation degree of structure and functions

The same procedure was applied for the estimation of the prospects of structure and functions, i.e., we estimated the ‘future degree of structure and functions’ at the sampling locality level, and then, we applied the rules presented in

Table 5 to estimate the future degree of structure and functions at the E.E.A. 10 km cell level and then the majority rule to assess the future prospects of habitat type, taking into consideration the prospects of the area and the range. The same procedure was also applied to estimate the conservation degree at the N2000 site, but in this case, only using the E.E.A. cells in which the site is included.

5 Discussion

5.1 Characteristics and advances of the proposed methodology

During the Greek monitoring project (2014–2015), we developed a methodological approach for the conservation degree/status assessment of structure and functions of habitat types based on quantitative methods (i.e., as objective as possible), which is, furthermore, flexible (e.g., leading to different types of analysis depending on the different scales), prone to optimization and improvement in its future implementations, and cost-effective, realistic, and as easy to use as possible (e.g., appropriate to be implemented by evaluators with a moderate level of specialization).

The first critical step towards the implementation of the described methodology was the classification of subtypes within habitat types, at least in the habitat types with considerable geographical and/or ecological differentiation in Greece, by means of numerical methods. In most cases, habitat types can be considered as distinguishable and repeatable assemblages of species (Henry et al. 2008). The distinction of different assemblages within habitat types representing their geographical and ecological variation is recommended by the explanatory notes and guidelines for the monitoring under Habitats Directive (Evans and Arvela 2011), as well as in the relevant literature (Henry et al. 2008; Kovač et al. 2016). The procedure which we have followed (classification of habitat subtypes and assignment of each sampling locality in a habitat subtype) ensures that species composition and abundance will be compared between ecologically and floristically similar vegetation types.

The second step was the selection of typical species for the identified habitat subtypes. The adopted methodology satisfies most of the characteristics that typical (indicator) species should have according to Carignan and Villard (2002). Specifically, we have chosen to use as typical species a rather long list of vascular plant taxa which can be considered as diagnostic of habitat types and their subtypes and were determined using objective methods. Subjectivity in choosing indicator species is considered as a primary limitation in their implementation (Siddig et al. 2016). The typical species chosen are the characteristic or differential taxa of the syntaxa (from the association to the class level) in which each habitat subtype is assigned. Such taxa may be considered as specialists, and may be sensitive to environmental changes and thus able to provide an early warning of changes (Carignan and Villard 2002; Hutto 1998).

The use of a long list of typical species (on average approximately 32 taxa) may be criticized, because it requires a lot of time during fieldwork (Bendali and Nellas 2016) and, in addition, it does not allow the collection of many detailed measurements (e.g., measurement of plants dimensions or of species densities). On the other hand, the use of a large number of typical species allows for more robust (based on larger samples) comparisons between areas or monitoring periods. Moreover, it ensures that various types of changes (e.g., response to management practices, environmental changes, and ecological succession) can be detected and that the causes of such changes can be investigated by means of appropriate analysis and ecological interpretation. Recording the complete floristic composition in each sampling locality would offer more opportunities to analyze the data, detect possible changes, and interpret the results (Yoccoz et al. 2001), but the choice to record only indicator species actually is a compromise to reduce the time and cost needed for field sampling, and to allow people with less floristic knowledge to contribute to field sampling. Actually, even citizen scientists could collect the relevant data, even though the citizen scientists are likely to detect less species in the field than the experts (Kallimanis et al. 2017).

Furthermore, the proposed method is not only based on the presence/absence of typical species, but it also includes additional information regarding their abundance and vitality. The estimation of the vitality of taxa is a fast but effective approach to assess if these species grow within their zone of ecological optimum or if they are subjected to stress. Presence/absence, abundance, and vitality are the main attributes of indicator species that are expected to reflect the cumulative effects of environmental changes (Siddig et al. 2016). Recording these attributes at the sampling locality allows for the synthesis–analysis of the data at higher spatial levels and thus capturing the multi-scalar patterns and processes of biodiversity (Rondinini and Chiozza 2010). Moreover, the record of presence/absence and abundance data for indicator species throughout Greece can also support the distribution modeling of the habitat types and thus resulting in the determination of their potential distribution. The latter is a useful parameter for the estimation of the habitat types' reference values concerning their area and range. Culmsee et al. (2014) pointed out that prediction of forest habitats is possible when it is based on indicator species with close association to these habitats.

Yoccoz et al. (2001) argue convincingly that the use of non-quantitative state variables in monitoring should be avoided. Thus, in the described approach, the conservation status assessment of the sub-parameter of typical species was based on the estimation of a continuous quantitative index (the ICDTS index). The latter index is useful to determine thresholds between the three different conservation degrees/statuses. Neither the Habitats Directive nor the explanatory notes and guidelines or any other publication answers to the question of how many of the typical species should be present with a favourable conservation status in an area to classify the conservation degree/status of a habitat type in a specific category (i.e., FV, U1, and U2). Furthermore, the surface of the area in which the above typical species should be present remains unspecified. The use of a continuous index helped us to search for thresholds and to deal with the uncertainty regarding the answer to the above-mentioned issues.

One critical issue about selecting indicators of structure and functions is the amount of effort and time needed to measure or estimate them during field sampling. We selected indicators which can be estimated relatively fast by evaluators that have undergone an appropriate training to attain a common understanding. Measurements of indicators such as those presented by Kovač et al. (2016) for forest habitat types (e.g., patch size, standing volume, and deadwood volume) would be surely more objective and accurate, but it would not be feasible and effective in terms of available time and budget to make such measurements in all (or in a large proportion) of the sampling localities. We consider that less accurate estimations in a considerable large number of sampling sites can reflect better the spatial variation of structure and functions of habitat types. In addition, they could be applied reliably by non-experts, like citizen scientists (Kallimanis et al. 2017).

The use of checklists for typical species, indicators of specific structure and functions, and pressures and threats alongside the application of numerical methods for the analysis of the collected data ensure that the final assessment will be independent—as much as possible—from the evaluator or the year of observation (Hill et al. 2005).

To test whether the proposed evaluation scheme was successful regarding its level of objectivity, we applied a Multiple Correspondence Analysis (MCA), an ordination analysis analogous to principal component analysis but for categorical variables (for more details, readers should refer to Borcard et al. (2011)). We included the outputs of assessment of each sampling locality (approximately 10,000 localities) in a database. More specifically, the assessment of (i) typical species, (ii) specific structure and functions, (iii) future trend, (iv) future status, (v) conservation degree (estimated at the N2000 site level), and (vi) head field evaluator was included in the database; and the sampling localities were grouped on the categories presented in Table 1. If the final assessment was highly subjective, one would expect that the 'conservation degree assessment (estimated at the N2000 site)' and the 'head field evaluator' would be located very close in the MCA plot.

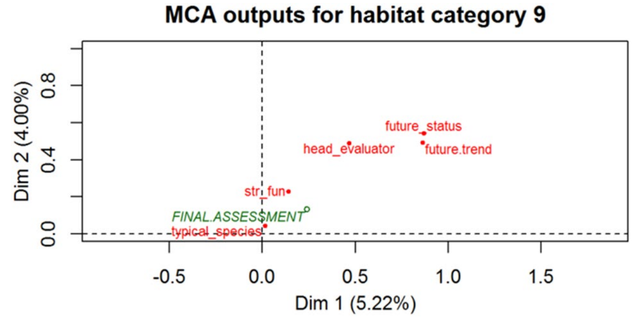


Fig. 5 MCA plot of riparian forest habitat types (habitat type category 9; Table 3). Variables: head_evaluator: head field evaluator of the sampling locality; FINAL_ASSESSMENT: conservation degree assessment of habitat type at the N2000 site level, future_status: assessment of future status; future_trend: assessment of future trend; str_fun: conservation degree of specific structure and functions, typical_species: conservation degree of typical species (colour figure online)

Instead, the conservation degree assessment was located closer to the variables 'typical species' and 'specific structure and functions', indicating that these had a stronger influence on the final assessment (Fig. 5).

The collected data using the proposed method can build a pressure-response scheme in the sense of Smeets and Weterings (1999), as they can be used for the description of the relationships between the origins and consequences of environmental problems. The data about the typical species as well as the specific structure and functions can be considered as indicators of biodiversity responses to environmental drivers, while the latter can be explored by analyzing the relationships between the indicators and the pressures/threats, as well as the environmental parameters collected in each sampling locality. According to de Bello et al. (2010), monitoring schemes should incorporate the assessment of both biodiversity indicators and drivers to be able to provide answers not only about the status of biodiversity, but also about the reasons behind this status. This is important for providing the appropriate data to policy makers to select effective measures to conserve biodiversity. Furthermore, the existence of data on both biodiversity indicators and drivers gives the possibility to develop modeling techniques for quantifying the effect of pressures and environmental factors on the conservation status of species and habitat types and thus to predict the conservation status under different scenarios of policy decisions (de Bello et al. 2010).

The application of a bottom–up evaluation approach provides the ability to analyze data at different spatial scales.

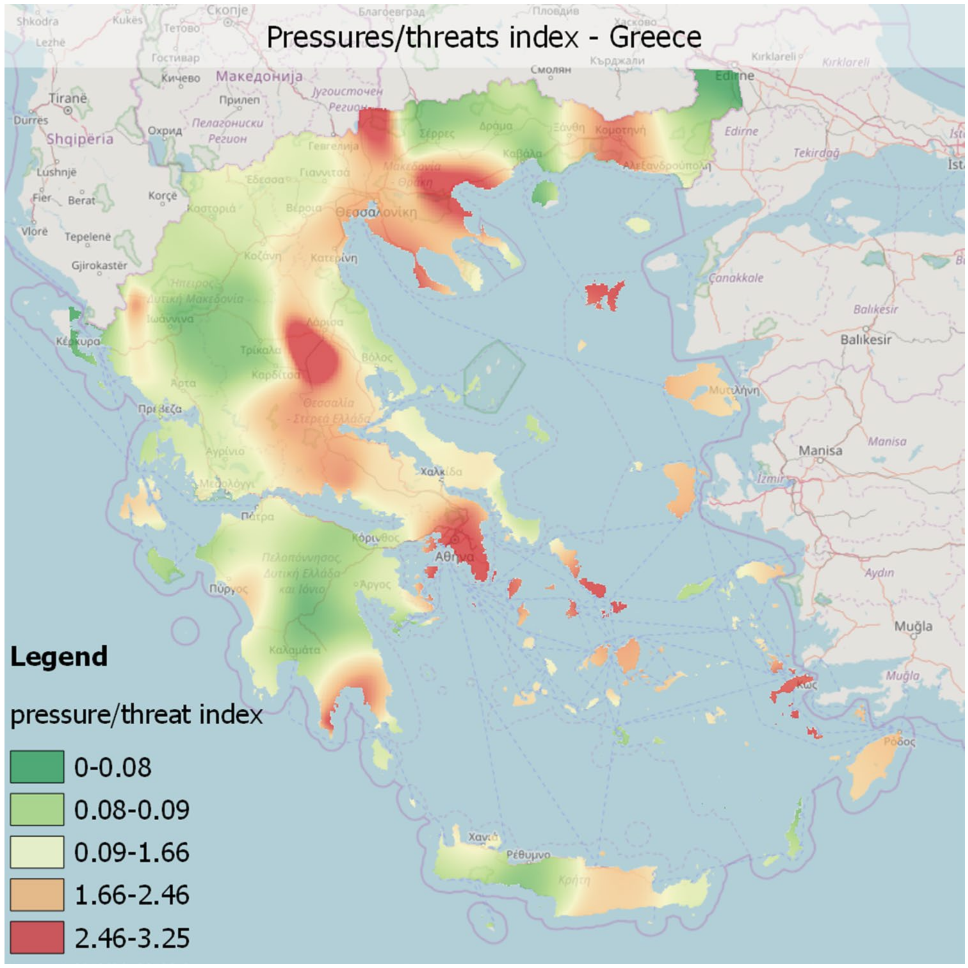


Fig. 6 Pressure/threat index values distribution over Greece interpolated (inversed distance weighting) from the mean pressure/threat index per sampling locality (colour figure online)

For instance, this approach results in specific knowledge at a local scale regarding the conservation degree or the occurrence of pressures and threats. This supports the elaboration of local management plans that are based on more localized or regional conservation objectives (Louette et al. 2011). To exemplify, the number and intensity of pressures and threats can be expressed in a pressure/threat index. This index is the transformation of the qualitative variable 'pressures and threats' into an ordinal scale. The transformation takes place using the following rules: A pressure/threat of low intensity obtains the value 1 in the ordinal scale, of medium intensity the value 2, and of high intensity the value 3. These are summed to the sampling locality level (if more than one pressure/threat has been recorded in the sampling locality) and then averaged to the E.E.A. grid level or directly interpolated over broader scales to represent areas with high values (many or intense) of pressures and threats (Fig. 6).

The bottom–up approach can partially counterbalance the negative effects of a spatially imbalanced sampling design through its potential to aggregate outputs using intermediate spatial scales, like the E.E.A. grid cell. Clustered sampling localities can distort the conservation status assessment if a disproportionally large number of sampling localities are located on small areas in which the habitat type exhibits a conservation degree which is different from that in extended areas that are under-sampled, when the aggregation of assessment is made directly from the sampling localities to the biogeographical region. Indications of irregular spatial arrangement of sampling localities should lead to the adoption of a stepwise aggregation procedure as it was done in Greece. For example, let us consider a habitat type that occurs in 6 N2000 sites (each being included in one E.E.A. 10 km grid cell), covering equal surface area in each site. In one of the sites, due to intense pressures and threats, the conservation degree of the habitat type is unfavourable-bad (U2), thus evaluators decide to sample 5 localities to depict the occurrence of various pressures and threats of high intensity. In the other sites, the habitat type is in favourable conservation status (FV) and field evaluators decide to sample one locality in each. Thus, for a total of ten sampling localities, five are evaluated as U2, and five as FV. Considering all sampling localities without further spatial aggregation, the conservation status assessment would result in U2 (more than

25% of sampling localities assessed as ‘U2’). A spatial aggregation in the E.E.A. 10 km grid cell would result in six grid cells, from which five would have been evaluated as ‘FV’ and only one (16%) as ‘U2’. In this case, the conservation status of the habitat type would result in ‘FV’, better representing the overall status of the habitat type in the country. This spatial aggregation provides some freedom to field evaluators to oversample localities in which the habitat type is facing problems without largely affecting the overall conservation status of the habitat type.

Among the major advances of the development of a standardized monitoring scheme is the enhancement of resources (both time and capital) efficiency of the monitoring project (Lengyel et al. 2008a). The use of site-based species’ checklists significantly restricts the time needed by evaluators to collect data in the field, and, additionally, decreases the required time for the identification of plant specimens in the laboratory.

5.2 Considerations and improvements needed

The distinguished habitat subtypes and the determination of their typical species are based on a database of approximately 18,000 vegetation plots. It is unknown if this data set represents the full range of variation (in terms of ecological conditions and floristic composition) of the habitat types occurring in Greece. Classification of vegetation data and determination of typical species should be repeated in the future and as long as a significant amount of new vegetation data will be available.

The thresholds for the classification of the typical species and the structure and functions in the three conservation statuses were chosen on the basis of best expert judgment. The use of indices to quantify the quality of structure and functions of habitat types is very useful (Kontula and Raunio 2009), but the choice of objective thresholds, especially for a high number of habitat types, is a very difficult task. More research is needed to this direction to establish ecologically meaningful thresholds concerning the typical species and the structure and functions. This possibly could be done by setting reference sites that represent different levels of conservation status (see Kovač et al. 2016). However, taking into account the ecological variability of habitat types, a high number of reference sites may be needed to find applicable thresholds. In addition, many habitat types are pioneer or intermediate stages of natural succession that have been developed after natural or human-caused disturbances (Halada et al. 2011). In such cases, the choice of reference sites is more difficult as they should represent certain ‘points’ in the continuum of ecological succession. Reference sites usually concern ecosystem types representing final stages of succession, with no or minimal disturbances. Another way to deal with the problem of threshold definition is an optimization procedure. In this case and through the accumulation of data, the already defined thresholds can be tuned to represent better the three conservation degrees. Furthermore, the optimization procedure should also concern the improvement of the lists of structure and functions, as well as of typical species. Although, for the sake of comparability between monitor periods, these parameters should be kept steady, a tuning of them would be most probably necessary on the basis of tests of their effectiveness to reflect habitat types conservations status as well as of new knowledge that would be produced (Yoccoz et al. 2001). In addition, the assessment of conservation degree of habitat types would be more complete if the lists of typical species included other groups of organisms (i.e., lichens, bryophytes, fungi, and animals) (Evans and Arvela 2011). More research is needed towards the identification of other groups of organisms that could be considered as indicators of the conservation degree of habitat types. An additional difficulty to integrate other groups of organisms in the typical species lists is that field researchers should have taxonomic and ecological knowledge for different groups of organisms.

6 Conclusions

In the frame of the Habitats Directive, all EU MS are obliged to establish a monitoring scheme and report the conservation status of habitat types and species of community interest every 6 years. This obligation should be taken as an opportunity to enhance the development of scientific methodology and research for biodiversity monitoring and conservation (Lengyel et al. 2008b) and to overcome the insufficient communication between scientific research, policy, and practice (Davis et al. 2014; Yoccoz et al. 2001). Monitoring programs aiming at the investigation of the effects of management actions to biodiversity surrogates can be considered as similar to the investigations of competing scientific hypotheses (Yoccoz et al. 2001). Here, we presented a methodology for the conservation status/degree assessment of habitat types structure and functions which (i) is largely based on quantitative methods, (ii) provides data that can be analyzed with different methods and at different spatial scales, (iii) is relatively cost-effective and realistic, and (iv) can and should be optimized in the future.

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References

Archaux F (2009) Could we obtain better estimates of plot species richness from multiple-observer plant censuses? J Veg Sci 20:603–611. https://doi.org/10.1111/j.1654-1103.2009.01079.x

Bendali F, Nellas N (2016) Conservation status assessment method for habitat types at Site of European Community Interest scale. Int J Innov Appl Stud 17:548–555

Bergstedt J, Westerberg L, Milberg P (2009) In the eye of the beholder: bias and stochastic variation in cover estimates. Plant Ecol 204:271–283. https://doi.org/10.1007/s11258-009-9590-7

Borcard D, Gillet F, Legendre P (2011) Numerical ecology with R. Springer Science & Business Media, New York

Braun-Blanquet J (1964) Pflanzensoziologie-Grundzüge der Vegetationskunde, 3rd edn. Springer Verlang, Wien

Buchanan GM, Butchart SHM, Dutson G, Pilgrim JD, Steininger MK, Bishop KD, Mayaux P (2008) Using remote sensing to inform conservation status assessment: estimates of recent deforestation rates on New Britain and the impacts upon endemic birds. Biol Conserv 141:56–66. https://doi.org/10.1016/j.biocon.2007.08.023

Cantarello E, Newton AC (2008) Identifying cost-effective indicators to assess the conservation status of forested habitats in Natura 2000 sites. Forest Ecol Manag 256:815–826. https://doi.org/10.1016/j.foreco.2008.05.031

Carignan V, Villard M-A (2002) Selecting indicator species to monitor ecological integrity: a review. Environ Monit Assess 78:45–61. https://doi.org/10.1023/a:1016136723584

Carli E, D'Alessandro E, Di Marzio P, Giancola C, Paura B, Salerno G, Blasi C (2016) Monitoring Natura 2000 habitats: habitat 92A0 in central Italy as an example. Biogeographia J Integr Biogeogr 31:7–25

Chrysopolitou V, Apostolakis A, Kotzageorgis G, Defingou M, Gioutlakis M, Hatziiordanou L, Hadjicharalambous H (2015) Deliverable B7. Methodology on data compilation for evaluating the habitat and species status, from cell level to Natura network and national level—final edition. YPEKA, Athens, Joint venture of Enveco S.A, Omikron Ltd., Karolidis Theodoros, Fyselias Spyridon, Consultant EKBY, Athens

Chytrý M, Tichý L, Holt J, Botta-Dukát Z (2002) Determination of diagnostic species with statistical fidelity measures. J Veg Sci 13:79–90. https://doi.org/10.1111/j.1654-1103.2002.tb02025.x

Culmsee H, Schmidt M, Schmiedel I, Schacherer A, Meyer P, Leuschner C (2014) Predicting the distribution of forest habitat types using indicator species to facilitate systematic conservation planning. Ecol Indic 37:131–144. https://doi.org/10.1016/j.ecolind.2013.10.010

Davis M, Naumann S, McFarland K, Graf A, Evans D (2014) Literature review, the ecological effectiveness of the Natura 2000 Network. ETC/BD Paris, France

de Bello F, Lavorel S, Gerhold P, Reier Ü, Pärtel M (2010) A biodiversity monitoring framework for practical conservation of grasslands and shrublands. Biol Conserv 143:9–17. https://doi.org/10.1016/j.biocon.2009.04.022

Del Vecchio S, Slaviero A, Fantinato E, Buffa G (2016) The use of plant community attributes to detect habitat quality in coastal environments. AoB Plants 8:plw040

Dierschke H (1994) Pflanzensoziologie: Grundlagen und Methoden. Ulmer, Stuttgart

European Commission (2006) Assessment, Monitoring and Reporting Under Article 17 of the Habitats Directive: Explanatory Notes and Guidelines

European Commission (2011) NATURA 2000 Standard Data Form Explanatory Notes

European Environmental Agency (2015) Results from reporting under the nature directives 2007–2012

Evans D, Arvela M (2011) Assessment and reporting under Article 17 of the Habitats Directive—explanatory notes & guidelines for the period 2007–2012—final Draft. European Topic Centre on Biological Diversity

Halada L, Evans D, Romão C, Petersen J-E (2011) Which habitats of European importance depend on agricultural practices? Biodivers Conserv 20:2365–2378. https://doi.org/10.1007/s10531-011-9989-z

Henry P-Y et al (2008) Integrating ongoing biodiversity monitoring: potential benefits and methods. Biodivers Conserv 17:3357–3382. https://doi.org/10.1007/s10531-008-9417-1

Hill D, Fasham M, Tucker G, Shewry M, Shaw P (eds) (2005) Handbook of biodiversity—survey, evaluation and monitoring. Cambridge University Press, New York

Hutto RL (1998) Using landbirds as an indicator species group. In: Marzluff JM, Sallabanks R (eds) Avian conservation: research and management. Island Press, Washington, DC, pp 75–92

Joint Nature Conservation Committee (2013) The UK Approach to Assessing Conservation Status for the 2013 EU Habitats Directive Article 17 Reporting. Peterborough

Kallimanis AS, Panitsa M, Dimopoulos P (2017) Quality of non-expert citizen science data collected for habitat type conservation status assessment in Natura 2000 protected areas. Sci Rep 7:8873. https://doi.org/10.1038/s41598-017-09316-9

Koleff P, Gaston KJ, Lennon JJ (2003) Measuring beta diversity for presence–absence data. J Anim Ecol 72:367–382. https://doi.org/10.1046/j.1365-2656.2003.00710.x

Kontula T, Raunio A (2009) New method and criteria for national assessments of threatened habitat types. Biodivers Conserv 18:3861. https://doi.org/10.1007/s10531-009-9684-5

Kovač M, Kutnar L, Hladnik D (2016) Assessing biodiversity and conservation status of the Natura 2000 forest habitat types: tools for designated forestlands stewardship. For Ecol Manag 359:256–267. https://doi.org/10.1016/j.foreco.2015.10.011

Lengyel S et al (2008a) Habitat monitoring in Europe: a description of current practices. Biodivers Conserv 17:3327–3339. https://doi.org/10.1007/s10531-008-9395-3

Lengyel S et al (2008b) A review and a framework for the integration of biodiversity monitoring at the habitat level. Biodivers Conserv 17:3341–3356. https://doi.org/10.1007/s10531-008-9359-7

Lennon JJ, Koleff P, GreenwooD JJD, Gaston KJ (2001) The geographical structure of British bird distributions: diversity, spatial turnover and scale. J Anim Ecol 70:966–979. https://doi.org/10.1046/j.0021-8790.2001.00563.x

Leuschner C, Ellenberg H (2017a) Ecology of Central European forests. Springer International Publishing, Switzerland

Leuschner C, Ellenberg H (2017b) Ecology of Central European non-forest vegetation: coastal to alpine, natural to man-made habitats. Springer International Publishing, Basel

Louette G et al (2011) Bridging the gap between the Natura 2000 regional conservation status and local conservation objectives. J Nat Conserv 19:224–235. https://doi.org/10.1016/j.jnc.2011.02.001

Maciejewski L (2010) Méthodologie d'élaboration des listes d' "espèces typiques" pour des habitats forestiers d'intérêt communautaire en vue de l'évaluation de leur état de conservation. Rapport SPN 12:57–58

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405:243–253. https://doi.org/10.1038/35012251

Nagendra H, Lucas R, Honrado JP, Jongman RHG, Tarantino C, Adamo M, Mairota P (2013) Remote sensing for conservation monitoring: assessing protected areas, habitat extent, habitat condition, species diversity, and threats. Ecol Indic 33:45–59. https://doi.org/10.1016/j.ecolind.2012.09.014

Rondinini C, Chiozza F (2010) Quantitative methods for defining percentage area targets for habitat types in conservation planning. Biol Conserv 143:1646–1653. https://doi.org/10.1016/j.biocon.2010.03.037

Schmeller DS (2008) European species and habitat monitoring: where are we now? Biodivers Conserv 17:3321–3326. https://doi.org/10.1007/s10531-008-9514-1

Siddig AAH, Ellison AM, Ochs A, Villar-Leeman C, Lau MK (2016) How do ecologists select and use indicator species to monitor ecological change? Insights from 14 years of publication in Ecological Indicators. Ecol Indic 60:223–230. https://doi.org/10.1016/j.ecolind.2015.06.036

Smeets E, Weterings R (1999) Environmental indicators: typology and overview. European Environment Agency Copenhagen, Copenhagen

Søgaard B et al (2007) Criteria for favourable conservation status in Denmark. Natural habitat types and species covered by the EEC Habitats Directive and birds covered by the EEC Birds Directive. National Environmental Research Institute—University of Aarhus

Spanhove T, Vanden Borre J, Delalieux S, Haest B, Paelinckx D (2012) Can remote sensing estimate fine-scale quality indicators of natural habitats? Ecol Indic 18:403–412. https://doi.org/10.1016/j.ecolind.2012.01.025

Tsiripidis I, Bergmeier E, Fotiadis G, Dimopoulos P (2009) A new algorithm for the determination of differential taxa. J Veg Sci 20:233–240. https://doi.org/10.1111/j.1654-1103.2009.05273.x/full

Vanden Borre J, Paelinckx D, Mücher CA, Kooistra L, Haest B, De Blust G, Schmidt AM (2011) Integrating remote sensing in Natura 2000 habitat monitoring: prospects on the way forward. J Nat Conserv 19:116–125. https://doi.org/10.1016/j.jnc.2010.07.003

Ward JH (1963) Hierarchical grouping to optimize an objective function. J Am Stat Assoc 58:236–244. https://doi.org/10.1080/01621459.1963.10500845

Yoccoz NG, Nichols JD, Boulinier T (2001) Monitoring of biological diversity in space and time. Trends Ecol Evol 16:446–453. https://doi.org/10.1016/S0169-5347(01)02205-4