



# Habitats on the grid: The spatial dimension does matter for red-listing



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## ABSTRACT

Besides species Red Lists, recently, a variety of frameworks have been proposed for assessing higher levels of biological organisation, i.e. ecosystems, habitats, plant communities. Most of these protocols refer to 'plant species assemblages' or 'vegetation types' as proxies for ecosystems or habitats. Indeed, the habitat concept based on plant communities has acquired a central role as a key approach for biodiversity conservation above the species level. Plant communities, like every complex biological system, hold scale-dependent 'emergent' properties which vary as a function of the scale of observation. With reference to red-listing, these scale-dependent properties have far-reaching consequences for both identification and classification, as well as for representation and evaluation, and become particularly challenging when dealing with criteria regarding decline in distribution or restricted distribution. The recent discussion on the red-listing protocols has evidenced several aspects that claim special efforts for a suitable use. In the present paper, starting with the analysis of some recently proposed protocols for the red-listing of habitats and ecosystems, we discuss and test some 'emergent' properties of species assemblages, providing cues for reflection. Based on a variety of theoretical models and scientific outcomes in literature from the last decades, we theorise that plant communities own some intrinsic, ecologically based and scale-dependent spatial features, which give rise to different types of pattern of spatial occupancy. We discuss a model where, in natural conditions, the possible patterns of spatial occupancy are referred to 3 basic types: areal, linear and point. This approach is here proposed as a tool to discriminate among different broad categories of plant community-based habitat types and optimise their assessment in the red-listing process. Starting from a homogeneous data set, the proposed case studies prove that the choice of the scale affects the comprehension of the habitats' occurrence, with a substantial relapse on the estimates of their distribution size. In particular, habitats with linear and point distribution, often naturally small in size and dispersed, are more susceptible to biased evaluation of their actual distribution and consequently of their threat status. The intrinsic spatial attributes of plant communities should not be neglected in a red-listing process and claim for a 'habitat-tailored' approach. The use of different grid-cell sizes and thresholds for the three main patterns of spatial occupancy here proposed, might certainly avoid inaccurate statements.

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## 1. Introduction

The conservation of biodiversity is a widely acknowledged target for humankind (Cafaro & Primack, 2014), as shown by an increasing array of regional, national and international agreements and frameworks for hindering biodiversity loss (see, e.g. CITES, 1973; European Commission, 2011; United Nations, 1976, 1992). The most challenging aspect stems from the complexity of biodiversity, that in itself comprises both multiple levels of organisation

(e.g. genes, species, communities and ecosystems) and their interacting relationships at all the integration levels (Allen & Starr, 1982; Margules & Pressey, 2000; Noss, 1990).

In the last decades, the International Union for Conservation of Nature (IUCN) has led the development of quantitative criteria for the creation of red lists of threatened species that allow for transparent, objective and repeatable risk assessments (IUCN, 2012, 2013; Mace et al., 2008). By ranking species at risk of extinction, the IUCN Red Lists provide a global indication on the state of one level of biodiversity and make governments and society aware of the trends in extinction risk (Baillie, Hilton-Taylor, & Stuart, 2004; Butchart et al., 2004; McCarthy, Thompson, & Garnett, 2008).

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Despite the cutting edge importance of species Red Lists, the realisation that an approach focused exclusively on the species level is unfit to conserve all components of biodiversity led the scientific community, conservation professionals and institutions to be increasingly concerned with biodiversity assessments, addressing higher levels of biological organisation (Izco, 2015; Keith, 2009; Keith et al., 2013, 2015; Kontula & Raunio, 2009; IUCN, 2015a; Nicholson, Keith, & Wilcove, 2009; Rodríguez et al., 2011, 2012, 2015). Ecological communities may more efficiently represent the biological diversity as a whole, compared to the species-level approach, which often lacks direct information about fundamental abiotic components, thus missing both the targets of protecting ecological patterns and processes, and ensuring the persistence of ecosystem functions and structure (Balmford et al., 2002; Cowling et al., 2004; Millennium Ecosystem Assessment, 2005; Noss, 1996; Secretariat of the Convention on Biological Diversity, 2010). Furthermore, the conservation of communities or ecosystems can also act as a surrogate for the species, particularly for those species yet undescribed or poorly known (Cowling & Heijnis, 2001; Nicholson et al., 2009), thus providing a precious service when considering that, despite strenuous efforts, only less than 5% of the estimated number of described species (less than 7% when considering only plants) has been evaluated for inclusion in the IUCN Red List by 2015 (IUCN, 2015b). The assessment of communities or ecosystems also allows to incorporate further information, such as the role of species richness/diversity, offering precious tools both for species and habitat's prioritization (see, e.g., Berg et al., 2014; Lindenmayer et al., 2008; Pärtel et al., 2005).

Early lists of endangered plant communities/habitats/ecosystems have been compiled since the 1980s, focusing either on specific ecosystem types or on national boundaries (see e.g., Moravec et al., 1983; Schulte & Wolf-Straub, 1986). Many common criteria were used for the evaluation (e.g. rarity, range, species composition, naturalness, human pressure, aesthetic or educational value), but the assessment was mostly based on expert knowledge, e.g. long-term field experience (Blab, Riecken, & Ssymank, 1995; Noss, LaRoe, & Scott, 1995; Paal, 1998) and, in some cases, key concepts were not underpinned by sound theoretical backgrounds. More recently, a variety of frameworks, founded on relevant ecological theories, have been proposed for assessing the threat status of plant communities/habitats/ecosystems (Berg et al., 2014; Biserkov, 2011; Essl, Egger, & Ellmauer, 2002; Lindgaard & Henriksen, 2011; Walker et al., 2006), in some cases prompted by government agencies. Nevertheless, differently from species, as yet there is no acknowledged international methodology on which to base the habitat red-listing. In Europe, Rodwell, Janssen, Gubbay, and Schaminée (2013) recently started a project aimed at developing a Red List of the European habitat types, while the IUCN council (CEM-IUCN & Provita, 2012) formally endorsed the protocol proposed by Keith et al. (2013).

Besides a common origin from the protocol developed for species (IUCN, 2013; Mace et al., 2008), the various methodologies adopted for plant communities/habitats/ecosystems red-listing share several characteristics: (a) although some protocols define assessment units of large dimensions, such as 'ecosystems' (Keith et al., 2013) or 'Land Environments' (Walker et al., 2006), most of them refer to 'plant species assemblages' or 'vegetation types', used as proxies for ecosystems or habitats; (b) as a rule, the assessment is based on quantitative criteria and only few protocols rely on qualitative ones, i.e. on the best expertise supported by specific paradigms (e.g. Biserkov, 2011); (c) all the protocols include decline and restricted size in spatial distribution as key criteria, and almost all identify quantitative thresholds, often analogous yet less severe than those adopted for species; (d) they are still rather lacking in suitable tools to incorporate measures of 'ecological function', i.e. the capacity of communities to support their whole

diversity of species and to sustain their functional roles in landscapes (Nicholson et al., 2009), with some remarkable exceptions giving special importance to species diversity or to the presence of threatened *taxa* in the evaluation of habitat quality (e.g. Andreas & Lichvar, 1995; Bacchetta, Farris, & Pontecorvo, 2012; Berg et al., 2014; Gauthier, Debussche, & Thompson, 2010).

### 1.1. Emergent properties: the "pattern of spatial occupancy"

Plant communities, like every complex biological system, hold aggregate or 'emergent' properties, which cause the whole to be more than the mere sum of its parts (Bissonette, 1997; Halley & Winkler, 2008; van der Maarel & Franklin, 2013). In natural conditions, properties of plant communities such as composition and structure arise from the interaction of both coarse- and fine-scale filters (Dale, 1999; Lortie et al., 2004). On a fine scale, vegetation patterns are ruled by species size and growth pattern, as well as by the interactions among plant individuals. On a larger scale, they are influenced by physical and geomorphologic features (i.e. valleys, ridges, slopes, water bodies) that create spatial and ecological heterogeneity (Dale, 1999; Greig-Smith, 1979; Palmer, 1988).

By interacting with spatially distributed environmental gradients, organisms, communities and ecological systems are thus arrayed in space to form distinct patterns or configurations, i.e. 'specific arrangement of spatial elements' (Turner, Gardner, & O'Neill, 2001), that exhibit a certain amount of predictability (Dale, 1999). Thus, although the boundaries between different plant communities are inherently more uncertain than is the case for species (Nicholson et al., 2009), the recognition and delimitation of stands of vegetation in the field can be based on internal characteristics, e.g. structural, physiognomic and floristic uniformity, and external ones, e.g. discontinuity with the surrounding vegetation (van der Maarel & Franklin, 2013).

On this ground, it can be assumed that each plant community owns intrinsic spatial features, ecologically founded, which affect its spatial distribution in natural conditions. In particular, the environmental heterogeneity accounts for an intrinsic property of each plant community that we call 'pattern of spatial occupancy' (PSO).

In accordance with Dale (1999), we refer to spatial pattern as 'nonrandomness in spatial arrangement, which then permits prediction'. The abiotic environment is spatially structured, resulting in patchy patterns or gradients. In natural conditions, when the plant species and assemblages are not subjected to human constraints, plant communities can display several, yet predictable, PSOs. Dale (1999) proposed the concepts of 'point pattern' and 'pattern on an environmental gradient' with reference to the type of representation of plant individuals and communities distribution. With different although closely related aims, the French phytosociological-synusial school developed a geometric approach to plant communities, introducing the shape ('forme spatiale, linéaire continue, linéaire discontinue, ponctuelle') as an essential property of vegetation patches in landscape analysis (Géhu, 1974, 1991; Julve, 1986).

As a general model, we assume that the PSO mostly tends to display three main patterns: *areal* (i.e. with an extended distribution; e.g. broadleaved temperate forests, natural and semi-natural grassland formations), *linear* (i.e. with a distribution in strips, where length is much greater than width; e.g. riparian and water-dependant formations, coastal plant communities) or *point* (i.e. with a naturally scattered spatial distribution, e.g. vegetation of temporary ponds).

By reflecting the ecological driving forces, these 3 models of PSO are representative of natural conditions and as such they should be considered as an intrinsic feature of each plant community (namely they do not give account of artificial, human-induced distribution, due e.g. to land-use change and fragmentation processes).

Properties of plant communities, such as composition and structure as well as their spatial pattern, are inherently scale-dependent concepts (Gray, Phan, & Long, 2010; Turner, O'Neill, Gardner, & Milne, 1989; Wiens, 1989, 1990). Scale dependence can be defined as the degree to which ecological phenomena vary as a function of grain and extent, the two main components of scale (Palmer & White, 1994). Since different patterns may emerge at differing scales of investigation, it is crucial that resolution (i.e. grain size) and extent be defined to accurately represent the ecological process or the object under study. Differently, the detected patterns will have little significance and will lead to inaccurate conclusions.

In a red-listing procedure, this issue becomes particularly challenging when dealing with criteria regarding declining or restricted distribution, i.e. traits based on spatial aspects of the unit under assessment. For instance, one of the worldwide accepted metrics used to evaluate the reduction in geographic distribution of any assessment unit is the 'Area of occupancy' (AOO), corresponding to the area of suitable habitat actually occupied by the assessment unit, usually calculated by crossing the occurrences with a grid and summing the areas (or counting the number) of the occupied cells (IUCN, 2012, 2013; CEM-IUCN & Provita, 2012). As such, AOO depends on the cell's size and is a function of the scale at which it is measured. Consequently, assessments are subjected to significant distortion if the distribution is assessed at different spatial scales. Given the general availability of spatial data, criteria based on distribution will likely be the most frequently used (Boitani, Mace, & Rondinini, 2015). In this sense, the development of consistent methods requires a careful investigation of the impact of the scale on the assessment outcomes.

Aim of the present paper is to show that the spatial pattern of plant communities, and in particular their PSO, might play a role in a red-listing procedure when applying quantitative criteria, specifically those relating to spatial occurrence. In particular, the use of the 'Area of occupancy' (AOO) might bring to inconsistent assessments across habitat types with different PSO. Our first hypothesis is that, as a direct consequence of the spatial distribution of ecological, physiognomic and structural variables, the PSO significantly affects the habitat's assessment. Our second hypothesis is that the use of a fixed grid-cell size for habitats with different PSO is not appropriate and leads to biased assessment, even more distorted for linear and point PSO.

## 2. Materials and methods

To show how the PSO might affect the AOO calculation and, consequently, the application of the red-listing criteria, we took into account four case studies.

As assessment units we considered the habitats as defined by the European 'Habitat' Directive 92/43/EEC (HD; see e.g. Evans & Arvela, 2011) and listed in its Annex I. The description of HD Habitats (from here on capitalised, to avoid any misunderstanding with other concepts of habitat), is founded on the related plant communities. They are usually identified, and even mapped, based on a phytosociological approach (Biondi, 2011; Braun-Blanquet, 1964; Dengler, Chytrý, & Ewald, 2008; Westhoff & van der Maarel, 1978), and are mostly described at the level of a syntaxonomic alliance (Biondi et al., 2012; Bunce et al., 2013; Evans, 2006, 2010; Rodwell et al., 2002). Being defined on the ground of their plant communities, HD Habitats can be assumed to own the same intrinsic, ecologically based, spatial features.

We took into account 126 Annex I Habitats occurring in Italy (Biondi et al., 2009). We used the data reported by the Italian Ministry of the Environment and Protection of Land and Sea for the 3rd report ex-Art. 17 (Genovesi et al., 2014; MATTM, 2013). More detailed data, derived from the Habitat maps included in Natura

2000 Sites Management Plans from the Regions Umbria, Veneto and Tuscany (unpublished data), have been used for a subset of 41 habitats.

To test our hypotheses, Annex I Habitats were grouped on the ground of their PSO. The adopted criteria for Habitat attribution to one of the three PSO (point, linear or areal) took into account, as main drivers, their intrinsic ecologic features, such as the presence of a strong ecological gradient, and the environmental spatial heterogeneity.

We considered as linear the riparian and coastal Habitats, plus all those Habitats which, for intrinsic reasons, when represented on a map appear in the shape of a stretch (e.g. rocky walls, steep slopes with chasmophytic vegetation). We considered as point those Habitats which, for intrinsic reasons, when represented on a map appear in the shape of spots, e.g., the temporary ponds, the inland salt meadows, the shallow standing oligotrophic waters, all characterised by a restricted distribution due to their ecological peculiarity. The Habitats whose distribution did not fall in the former two types have been referred to the areal PSO. The composition of the groups is reported in Table 1; for Habitat descriptions see Biondi et al. (2009, 2012).

To test if different Habitats, grouped according to their PSO, exhibit significant differences in the average spatial extent, firstly we considered a theoretical case study (a), represented by three Habitats with the same actual surface but different types of PSO, and their interrelation with a  $10 \times 10 \text{ km}^2$  grid, on whose base the AOO has been calculated.

Then, in the case study (b), we analysed the average total surface per Natura 2000 site occupied by each of the considered 126 Annex I Habitats, mapped and measured all over the Italian Natura 2000 network (2585 sites).

In case study (c) data on the actual surface ( $\text{km}^2$ ) have been related to the calculated AOO (based on a  $10 \times 10 \text{ km}^2$  grid), with the aim to check if the 3 PSO-based groups of Habitats evidenced any differential response. In order to normalise data, we compared the ratio between the total grid-based AOO surface and the total actual surface ( $\text{km}^2$ ) for each Habitat. This metrics gives also account on how large is the gap between the two considered values.

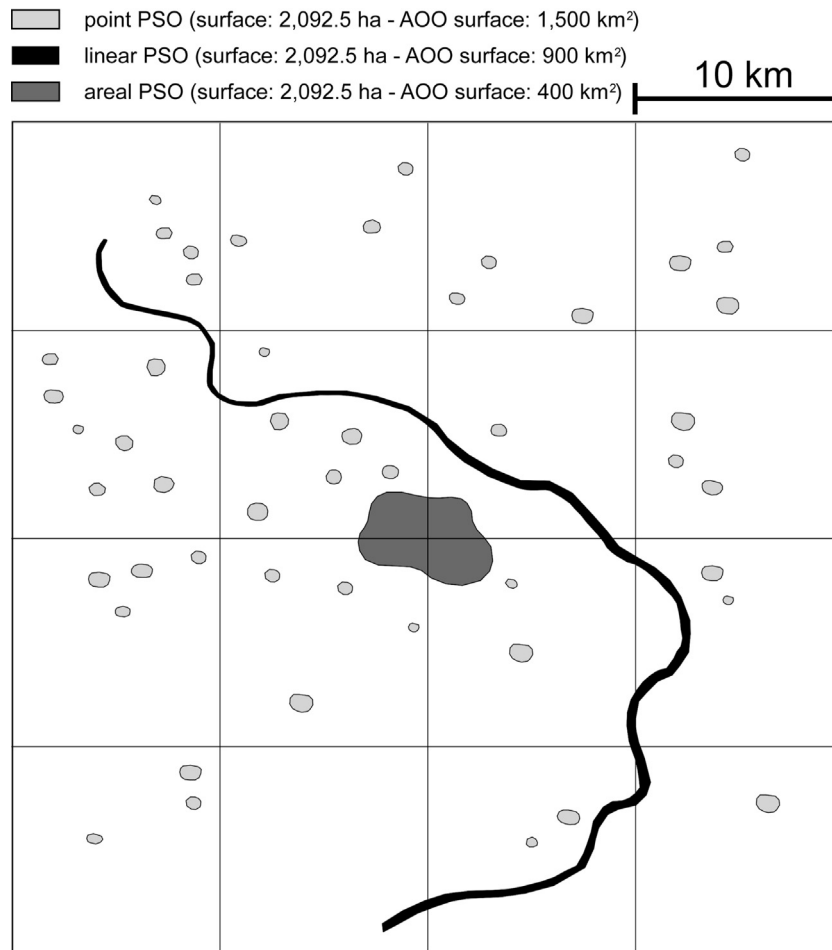
Eventually, in the case study (d), we applied the AOO calculation process to a restricted number (41) of Habitats present in Italy, for which detailed cartographic data were available, with the aim to detect if the AOO calculation is affected by the PSO and by the adopted grid cell-size. The 41 Habitat types used for case study (d) are included in the complete list (Table 1), with the indication of the territorial scale they have been considered at (national, regional or local). For each considered Habitat type, the detailed cartographic data allowed to calculate the AOO surface at four different grid-cell sizes ( $10 \times 10 \text{ km}^2$ ,  $2 \times 2 \text{ km}^2$ ,  $1 \times 1 \text{ km}^2$  and  $0.5 \times 0.5 \text{ km}^2$ ).

For all the case studies, the statistical significance of the differences among the considered groups of data has been analysed by applying the nonparametric Kruskal-Wallis Test. The correlation analyses have been based on the non-parametric Spearman's rank coefficient. The software Analyst Soft StatPlus: mac v2009 has been used for statistical analyses. The cartographic data have been processed by using the Open Source Geographic Information System Quantum GIS QGIS 2.0.1 (QGIS Project, 2014).

## 3. Results

### 3.1. Case study (a): a simple exercise of 'form'

The three theoretical Habitats sharing the same surface (2092.5 ha) and differing in the type of PSO (areal, linear and point), are spatially represented in Fig. 1. They might correspond, e.g., to a dry grassland (Annex I code: 6210), a *Salix alba*-dominated riparian



**Fig. 1.** Exemplificative representation of three theoretical habitats with the same actual surface (2,092.5 ha) but different types of PSO, and their interrelation with a  $10 \times 10 \text{ km}^2$  grid; the respective calculated values of the AOO surfaces are reported (map processed by using QGIS 2.0.1).

gallery (92A0) and a Mediterranean temporary pond with *Isoëtes hystrix* (3170), respectively. The calculated AOO surface, based on a  $10 \times 10 \text{ km}^2$  grid, resulted respectively in 400, 900 and  $1500 \text{ km}^2$ .

### 3.2. Case study (b): ranges of size of Habitats: scale and grain play a role

Results of the comparison among the average total surface of Annex I Habitats per Natura 2000 site, grouped according to their PSO, are shown in Fig. 2. The mean Habitat surface per site showed a decreasing trend from areal (311.8 ha), to linear (79.5 ha), to point Habitats (28.0 ha), resulting in statistically significant differences ( $p < 0.001$ ) among the three PSO types, on a wide (national) scale. This indicates that the three PSO-based groups of Habitat types show distinct ranges of size and ask for own peculiar scales of representation.

### 3.3. Case study (c): a theoretical exercise of form applied to actual distribution data

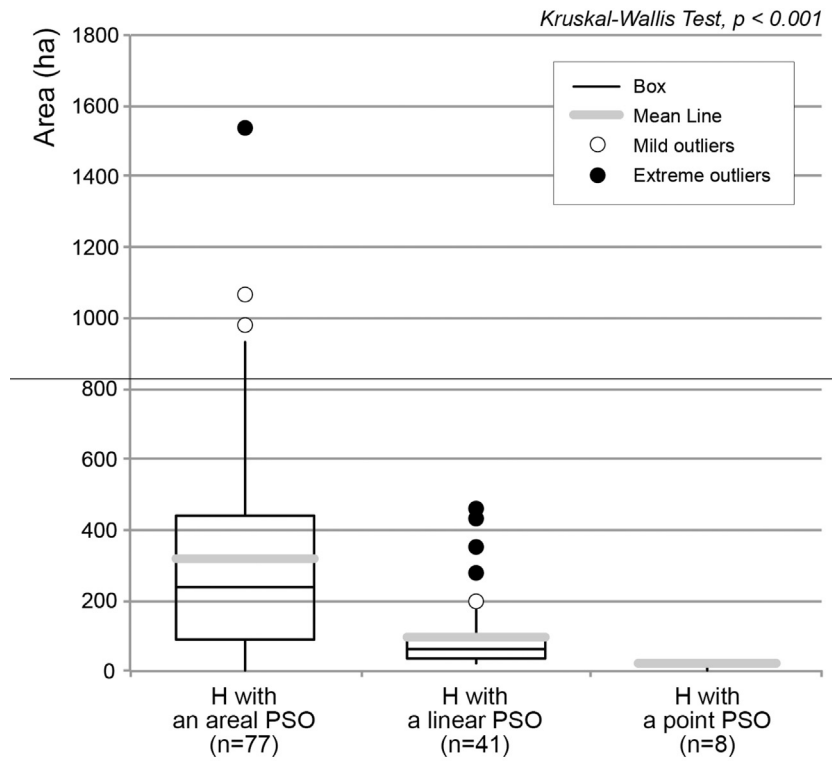
Results of the analysis on the relation between the calculated AOO and the actual surface of each Habitat type are shown in Fig. 3.

Both Habitats with a point and a linear PSO were heavily shifted in favour of inflated values of the grid-based surface (respectively:  $R = 0.952$ ,  $p < 0.001$ ;  $R = 0.759$ ,  $p < 0.001$ ). Conversely, Habitats with

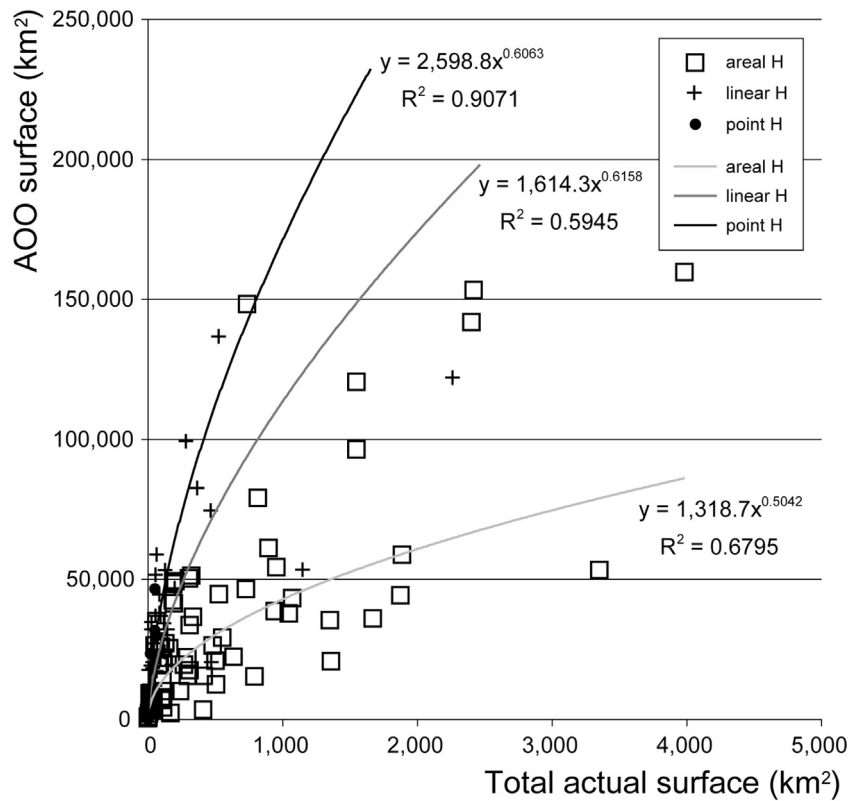
an areal PSO showed a clear tendency to correlate with proportionally lower values of the AOO surface ( $R = 0.807$ ,  $p < 0.001$ ).

### 3.4. Case study (d): effects of grid size and pattern of spatial occupancy on AOO

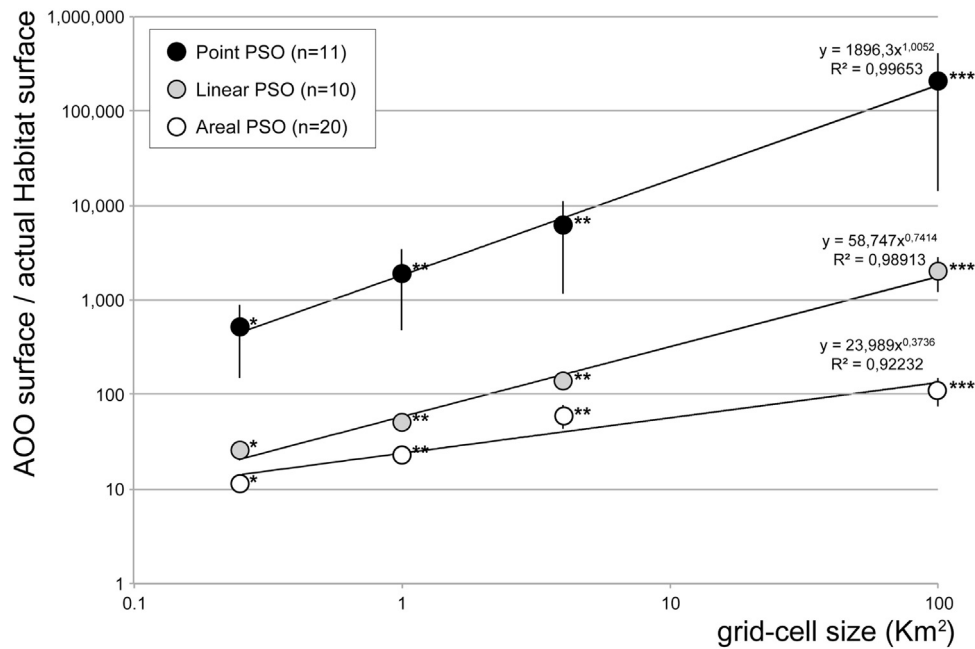
The effect of AOO based on a fixed grid-cell size for Habitats with different PSO is shown in Fig. 4. The y axis shows the ratio AOO/actual surface, for each PSO-based group of Habitats (average values per group are represented); the x axis shows four series of data, referring to the four grid-cell sizes used for the AOO calculation ( $10 \times 10 \text{ km}^2$ ,  $2 \times 2 \text{ km}^2$ ,  $1 \times 1 \text{ km}^2$  and  $0.5 \times 0.5 \text{ km}^2$ ). For point Habitats (and, at a lesser extent, for linear Habitats) the gap between AOO and actual surface was always remarkably higher than for areal habitats. This gap largely increased when shifting from a small- to a larger-sized grid. Overall, a large grid-cell size produced high AOO estimates, while a smaller size produced AOO surfaces that better approximated the real surfaces (Spearman's correlation, respectively:  $R = 0.952$ ,  $p < 0.001$ ;  $R = 0.759$ ,  $p < 0.001$ ;  $R = 0.807$ ,  $p < 0.001$ ). Furthermore, if we consider the trendlines, which represent how the ratio AOO/actual surface increases with the progressive enlargement of the grid-cell size, a further insight becomes evident: the inflation is significantly more pronounced when considering Habitats with a point or linear PSO compared to an areal PSO, showing differences of several orders of magnitude among them.



**Fig. 2.** Average Annex I Habitat (H) surface (ha) per Natura 2000 Site in Italy, with Habitats grouped according to their PSO: point, linear or areal; elaboration based on original data from 2585 SCI and/or SPA from Italy; the statistical significance of the differences among groups is reported, based on Kruskal-Wallis Test (Data Source: [MATTM, 2013](#)).



**Fig. 3.** Relations between the AOO surface (based on a  $10 \times 10 \text{ km}^2$  grid) and the total actual surface ( $\text{km}^2$ ) for 126 Habitats occurring in Italy; symbols and trendlines are referred to the three PSO-based groups of Habitats (Data Source: [MATTM, 2013](#)).



**Fig. 4.** Ratio between the AOO surface (calculated with progressively larger grid-cell size:  $0.5 \times 0.5$ ,  $1 \times 1$ ,  $2 \times 2$ ,  $10 \times 10$  km<sup>2</sup>) and the actual measured surface, for 41 Italian Annex I Habitat types for which exhaustive analytic data were available at different scales. Habitats are grouped according to the 3 models of PSO; bars: standard error; the statistical significance of the differences among groups is reported, based on Kruskal-Wallis Test: \* $p < 0.05$ ; \*\* $p < 0.01$ ; \*\*\* $p < 0.001$  (Data sources: Natura 2000 Management Plans; MATTM, 2013).

#### 4. Discussion: lessons learned and perspectives

The results of the four case studies help pointing out some issues that might be worth thinking and discussing. As main outcome, results highlight the prominent role of scale and grain when quantifying the distribution range of a Habitat and of any plant community-based habitat type.

The importance of the scale has been widely recognised as it concerns all types of ecological data and is a fundamental aspect of the environmental heterogeneity, whose interpretation depends on the level of observation established when studying an ecological system (Battisti & Fanelli, 2015; Levin, 1992; O'Neill et al., 1991, 1996).

In theory, any scale can be used to study ecological problems, and no specific spatial scale may be universally applied; nevertheless, the scale at which environmental patterns are quantified influences the result (Brandt, 1998; Turner et al., 2001; Wiens, 1989; Wu, 2004) and inappropriate scales may fail in detecting patterns (Li & Wu, 2004; Kirchheimer et al., 2015). With increasing grain size (and decreasing scale), units which are intrinsically rare or naturally occur as very small stands may become less represented or even disappear (Turner et al., 2001). This is generally true when distribution maps based on polygons are used. But it is true as well when adopting the AOO as a proxy for distribution. Estimates of AOO appear to be highly sensitive to spatial grain, as also suggested by other Authors (see e.g., Hartley & Kunin, 2003; Nicholson et al., 2009). The surface (and the number of cells) corresponding to the AOO obviously depends on the cell size and is a function of the scale at which it is measured.

The here discussed case studies show that the AOO scale-dependence is also strongly affected by an emergent property of plant communities: the PSO, an inherently scale-dependent concept. With reference to red-listing, scale-dependent properties have far-reaching consequences for both identification and classification, as well as for representation and evaluation. Inconsistency in the classification and mapping scale of ecological communities

may lead to inaccurate statements and affect the outcomes of the assessment (Nicholson et al., 2009).

The case study (a) showed the dramatic effect of the PSO on the AOO calculation: the PSO affects the number of units of occurrence, regardless of the grid-cell size. We used a grid with  $10 \times 10$  km<sup>2</sup> cell size, as recommended by Keith et al. (2013), but similar results would be obtained with other theoretical habitats and grids with larger or smaller cells. Although sharing the same total distribution surface, the AOO of the 3 theoretical Habitats drastically varied. The inappropriateness of AOO becomes particularly apparent for those Habitats (and related plant communities) whose natural PSO is the result of strong environmental driving forces, which determine peculiar geometries in their space occupancy. It might be the case, for instance, of the vegetation colonising the land immediately adjacent to a river, the so called 'riparian zone', a definition that accounts for its more or less narrow and linear distribution (Cressie, 1993; Diggle, 1983; Malanson, 1993; Upton & Fingleton, 1985, 1989). Similarly, the intrinsic fragmentation of a point Habitat type will always result in a remarkable inflation of its AOO, which in Fig. 3 is one order of magnitude higher than the areal one, due to its scattered distribution.

The three models of PSO adopted in our analysis show, with a simple approach, that the intrinsic spatial patterns of plant communities should not be neglected when assessing their distribution size. Starting from a homogeneous data set, the proposed case studies proved that the choice of the resolution scale affects the comprehension of the patterns of space occupancy and occurrence of the Habitats and the correlations between Habitats and environment (Greig-Smith, 1983; Juhász-Nagy & Podani, 1983; Reed, Peet, Palmer, & White, 1993), with a substantial relapse on the measurement of their AOO.

Case study (c) demonstrated that the use of a common grid for the AOO calculation affects the outcomes in a differential way for the 3 PSO types, generating inconsistent results. Habitats with a point or linear PSO are those with a major risk to be over-estimated when relying on AOO, while for those with an areal PSO the AOO surface results far closer to the actual distribution area. Thus, as

**Table 1**

Attribution of the Italian Annex I Habitats to the 3 PSO.

List of the Annex I Habitats occurring in Italy, with the related codes (data sources: MAITM, 2013), considered in the data analyses, grouped on the ground of their pattern of spatial occupancy (PSO); the letters in brackets 'N', 'R' and 'P' indicate the Habitat types selected for case study (d) and report the territorial range the data refer to (respectively: national, regional, provincial).

Habitats with an areal PSO
<ul style="list-style-type: none"> <li>• Open sea and tidal areas: 1110, 1120, 1130, 1140, 1150, 1160, 1170</li> <li>• Temperate heath and scrub: 4030, 4060, 4070, 4080, 4090</li> <li>• Sub-Mediterranean and temperate scrub: 5110, 5130</li> <li>• Mediterranean arborescent matorral: 5210, 5230</li> <li>• Thermo-Mediterranean and pre-steppe brush (except 5320 because related to the gradient of the littoral systems): 5330</li> <li>• Phrygana: 5410, 5420, 5430</li> <li>• Natural grasslands: 6110, 6130, 6150, 6170</li> <li>• Semi-natural dry grasslands and scrubland facies: 6210, 6220, 6230, 6240, 62A0</li> <li>• Sclerophyllous grazed forests (dehesas): 6310</li> <li>• Molinia meadows on calcareous, peaty or clayey-siltladen soils (Molinion caeruleae): 6410</li> <li>• Mesophile grasslands: 6510, 6520</li> <li>• Sphagnum acid bogs: 7110 (N,R), 7120, 7140 (P), 7150</li> <li>• Alkaline fens: 7230 (R,P)</li> <li>• Alpine pioneer formations of <i>Caricion bicoloris-atrofuscae</i>: 7240</li> <li>• Scree: 8110, 8120, 8130</li> <li>• Siliceous rock with pioneer vegetation of the <i>Sedo-Scleranthion</i> or of the <i>Sedo albi-Veronicion dillenii</i>: 8230</li> <li>• Limestone pavements: 8240</li> <li>• Fields of lava and natural excavations: 8320</li> <li>• Permanent glaciers: 8340</li> <li>• Forests of temperate Europe (except 91B0, 91E0, 91F0 because depending on the presence of water gradients): 9110 (R), 9120, 9130, 9140, 9150, 9160, 9180 (R), 9190, 91AA, 91D0, 91H0, 91K0, 91L0 (R), 91M0 (R)</li> <li>• Mediterranean deciduous forests (except 92A0, 92C0, 92D0 because depending on the presence of water gradients): 9210 (R), 9220, 9250, 9260 (R)</li> <li>• Mediterranean sclerophyllous forests: 9320, 9330, 9340 (N), 9350, 9380</li> <li>• Temperate mountainous coniferous forests: 9410 (R), 9420, 9430</li> <li>• Mediterranean and Macaronesian mountainous coniferous forests: 9510, 9530, 9540, 9560, 9580, 95A0</li> </ul>
Habitats with a linear PSO
<ul style="list-style-type: none"> <li>• Sea cliffs and shingle or stony beaches: 1210, 1240</li> <li>• Salicornia and other annuals colonising mud and sand: 1310</li> <li>• <i>Spartina</i> swards (<i>Spartinion maritimae</i>): 1320</li> <li>• Mediterranean and thermo-Atlantic saltmarshes and salt meadows: 1410, 1420, 1430</li> <li>• Salt and gypsum inland steppes: 1510</li> <li>• Sea dunes of the Atlantic, North Sea and Baltic coasts: 2110, 2120 (R), 2130 (R), 2160</li> <li>• Sea dunes of the Mediterranean coast: 2210, 2230, 2240, 2250, 2260, 2270</li> <li>• Hard oligo-mesotrophic waters with benthic vegetation of <i>Chara</i> spp.: 3140</li> <li>• Natural eutrophic lakes with <i>Magnopotamion</i> or <i>Hydrocharition</i> – type vegetation: 3150</li> <li>• Running water: 3220, 3230, 3240 (R), 3250, 3260, 3270, 3280 (R), 3290</li> <li>• Low formations of <i>Euphorbia</i> close to cliffs: 5320</li> <li>• Semi-natural tall-herb humid meadows (except 6410): 6420, 6430 (R)</li> <li>• Calcareous fens (except 7230, 7240 which have an areal distribution): 7210 (R), 7220 (P)</li> <li>• Calcareous rocky slopes with chasmophytic vegetation: 8210 (R)</li> <li>• Siliceous rocky slopes with chasmophytic vegetation: 8220 (R)</li> <li>• Forests of temperate Europe depending on water or impluvia: 91B0, 91E0 (R), 91F0 (R)</li> <li>• Mediterranean deciduous forests depending on water or impluvia: 92A0 (R), 92C0, 92D0 (R)</li> </ul>
Habitats with a point PSO
<ul style="list-style-type: none"> <li>• Inland salt meadows: 1340 (N)</li> <li>• Inland dunes with open <i>Corynephorus</i> and <i>Agrostis</i> grasslands: 2330 (N)</li> <li>• Standing water (except 3140, 3150): 3110, 3120, 3130, 3160, 3170 (R)</li> <li>• Submerged or partially submerged sea caves: 8330</li> </ul>

evidenced in case study (d), the appropriateness of a grid-cell size for the AOO calculation is deeply contingent on the Habitat PSO, and Habitat types with a point or linear PSO will always risk to be overestimated when their AOO is calculated based on a common grid-cell size.

The issue has already been faced with reference to species red-listing. Many authors agreed in recognising that different spatial scales of observation can reveal different threshold patterns for different species (Forman, 1995; Huggett, 2005; Mac Nally et al., 2002). The risk of inconsistencies and bias caused by the AOO estimation at different scales was already emphasised by IUCN for species, with reference to specialised *taxa* with a patchy distribution, evidencing the need for scale correction factors (IUCN, 2012, 2013). IUCN even approached the problem of species with 'linear' habitats, eventually suggesting to neglect this peculiar ecological distribution, since in such cases the habitat actual surface might be so narrow that it should nearly always be assigned to a threatened status, based on the 'B' criterion (IUCN, 2013).

Nevertheless, there is a general agreement on the recognition of intrinsic (specific) needs of different species with reference to pressures and extinction risk (Johnson, 2013).

Dykstra (2004) reported that there are no general response models in species decline, making the identification of generally valid thresholds particularly hard. Indeed, a generalisation is hard to find for single species, due to the enormous variety of ecological requirements and traits, including size and specialisation. Similarly, it is practically impossible to adopt specific rules for each single Habitat. However, some intrinsic spatial properties of each plant community-based habitat type (e.g. the here proposed PSO) might represent a good model and offer a major advantage in that they allow to take into account, at least in part, some prominent community attributes and to modulate not only metrics but also thresholds and criteria of assessment. Additionally, it is perhaps worth to consider that, although AOO is one of the most used metrics to assess the conservation status of a habitat, it should not be used alone, but should be only one in a set of proxies.

IUCN (2012) recommends to use a scale that is 'appropriate to relevant biological aspects' of the assessment unit. In this light, the proposal by Keith et al. (2013) to calculate AOO on the base of a standardised spatial grain, recommending a  $10 \times 10 \text{ km}^2$  grid and fixing common quantitative thresholds for all the assessed items, appears rather difficult to understand, if not even lacking a clear theoretical basis, as suggested by Boitani et al. (2015).

Our findings highlight the need of a certain adaptability in the red-list criteria when assessing ecological communities, with particular reference to their spatial dimension, in accordance with Nicholson et al. (2009) who argued that communities should be listed at the broadest scale at which they meet criteria for listing. Habitat types with a linear or a point PSO, naturally small in size, scattered or with a stretched distribution, need a finer-scale approach to be accurately identified and correctly assessed, as already emphasised for species (see e.g., Jiménez-Alfaro, Draper, & Nogués-Bravo, 2012).

Although it is beyond the scope of this study, the reported case studies concur to confirm that plant communities are suitable operational objects in the red-listing process. Despite their great complexity, plant communities and plant community-based habitat types are organized in the form of spatially recognisable patches. They appear to be unequivocal assessment units, better defined compared to the concept of 'ecosystem' used by Keith et al. (2013), as already indicated by Boitani et al. (2015). There is a huge convergence between plant communities, HD Habitats and other officially acknowledged vegetation classification systems, such as EUNIS (Davies, Moss, & Hill, 2004) or CORINE biotopes (Commission of the European Communities, 1991), as clearly exemplified by the European Commission (2013). This explains why in Europe there

are many efforts to find an interoperability between existing classifications of habitats and plant communities (see, e.g., Rodwell et al., 2002; Schaminée et al., 2012, 2013a, 2013b).

#### 4.1. Conclusions

With the present paper we aim to advance some viewpoints that might consolidate the grounds but also open new perspectives for the increasing interest in the assessment of higher-than-species biodiversity levels. Facing the risk of unreliable assessments, some possible solutions should go in the following directions: (i) the actual distribution surface of a Habitat should always be seen as the optimal proxy to take into account; (ii) grids should only be limited to cases where the level of knowledge is really low; (iii) when using grids, the cell size (or, possibly, the threshold values for different threat categories) should be selected based on the PSO of the assessment unit.

There is substantial scope for exploring the spatial dimension of Habitats and future studies should implement researches specifically designed to address this issue. It is indeed a central point for red-list assessments and highlights what we believe to be some novel future direction for this field.

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