A STANDARDIZED PROCEDURE FOR SURVEILLANCE AND MONITORING EUROPEAN HABITATS.

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with

ORIGINAL CONTRIBUTION: A NEW DESCRIPTOR OF PLANT COVER SYNGENESIS – DEGREE OF NATURALNESS AS A MANAGEMENT QUALIFIER.

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Abstract

Presented paper is composed of two parts: the first represents an extract from the collective publications (Bunce *et al.* 2005, Bunce *et al.* 2007) with the aim to transfer the concept to the Czech public and territory, the second one (on a new management qualifier) is original and firstly published with reference to oral presentation by the author within the BioHab Prague workshop (Kovář 2004b). BioHab (Biodiversity and Habitats) represents the project aiming to creation of user-friendly system for surveillance and monitoring of European habitats. Well defined scientific and policy requirements for a practical, transmissible, and reproducible procedure are known. A procedure is described that will satisfy these requirements and which can provide the necessary data that are currently lacking (Bunce *et al.* 2005; 2007). Rigorous rules are given: the procedure is based on plant life forms, used in biogeography since the nineteenth century, and on the underlying statistical correlation between these life forms and the environment. This relationship has been validated statistically and the procedure has also been tested in the field in all European environmental zones. There are distinguished 130 General Habitat categories. They are enhanced in the field by recording environmental, site and management qualities to enable flexible database interrogation. The same categories are applied using rules, with appropriate qualifiers, to areal, linear and point features, so that for the first time integrated reporting of variation of habitats at the landscape level can be carried out. It therefore incorporates landscape ecological principles such as landscape diversity and will enable both connectivity and fragmentation to be assessed at the detailed landscape level.

Degree of plant stand similarity as an expression of different naturalness/syngenesis indicated by Jaccard index is demonstrated as a tool for description of management history. This new management qualifier can be used especially in territories of good phytosociological traditions in vegetation science with experienced applications into habitat classifications and land use planning.

Key words: landscape-ecological sampling framework, field recording, surveillance, monitoring, habitat, biodiversity, life forms, management qualifier, degree of naturalness, plant cover syngenesis, Jaccard index of similarity

Introduction

More than three last decades, various ways of field recording and measuring of lansdcape-ecological features have been tested for integration using strategic sampling framework. Monitoring is needed not only for NATURA 2000 sites, but also for other initatives, such as indicators of sustainable development and biodiversity action plans. Not only consistent data to answer these requirements, but also a standardised rule based procedure is not available. The approach described in the present work fills this gap and is based on the BioHab project carried out in the EU Fifth Framework programme (EVK2-CT-2002-20018). Whilst the core of the procedure concerns rules and instructions for consistent field recording, it is essential that they are linked to a spatial framework for the whole of Europe. Such a framework is therefore integral to the methodology as it provides a means to extend the detailed samples needed to assess habitats in the field to European estimates. The temporal dimension is added by describing the monitoring procedure. Land managers demand indicators of direct relationships between biodiversity and habitats as well as historical trends of ecological changes. Responsibility of man in driving global environmental change make the demand for its evidence and relevant statistics are not only important for local and national policies, but are also used to evaluate international conventions (e.g. the Gothenberg Commitment by the European Union to stop biodiversity loss by 2010). Unfortunately, there is a lack of consistent data to meet this requirements, especially for assessments at a supra-national level, where (within the EU: under the Habitats and Species Directive – European Communities 1992) countries are obliged to monitor and assess changes in biodiversity and habitats. Projects such as MIRABEL (Petit *et al.* 2001) have only been able to use expert judgement for assessing the distribution and extend of European habitats. Műcher *et al.* (2004) have tried to derive rules using existing databases to predict distribution of habitats. However, many of the descriptions do not contain enough detail for mapping. This is the reason why the BioHab project brings a procedure that has been developed to enable statistically rigorous estimates in the territorial frame of European habitats, based on consistent field recording of habitat variability (incl. points, lines and patches) using reproducible and transmissible rules.

Field recording has been at the core of ecology since its inception as a recognisable science. Long-term monitoring is more common in species oriented approaches e.g. birds and carabid beetles (Den Boer et Van Dijk 1994). The development of vegetation science as part of ecology has been mainly descriptive and based on the selection of homogenous stands of vegetation, usually relatively undisturbed (Braun-Blanquet 1928; Tüxen 1937). Such work is not designed for long-term monitoring, although the individual records are suitable, if they are relocateable (e.g. Grabherr *et al*. 1994). Long-term vegetation monitoring studies are restricted. For example, in the UK (Bunce *et al*. 1993) found under five case studies of long-term vegetation monitoring. Similarly, quantitative ecologists, e.g. Greig-Smith (1964) were largely concerned with technical, as opposed to actual practical problems. Bunce et Shaw (1973) described a standardised procedure, which was applied to British woodlands in 1971 and subsequently repeated in 2003 (Kirby *et al*. 2005) demonstrating that statistical rigour is essential for long-term monitoring. A stratification of framework has been constructed that optimises the selection of sampling locations. Some experiences (Bunce *et al.* 1996) have used independent environmental classifications derived from existing biogeoclimatic information. This approach has been developed also in Spain (Elena Rossello 1997) or e.g. in Australia (Cawsey *et al.* 2002) at a continental scale. However, all these approaches do not involve recording at the landscape level i.e.

involving complexes of habitats, as described by Sheail et Bunce (2003) and Fjellstad *et al.* (2001). Any procedure must recognise and utilise this complexity – hence the development of the present approach. In contrast, most European approaches to the assessment of habitats either ignore these requirements, or deliberately avoid landscape complexity by selection of sites that are considered homogenous. In the 1950's and 60's, the development of the ecosystem concept was restricted, albeit not explicitly, to concepts of vegetation classification (comparison at various methodological levels were published later, e.g. Kovář et Lepš 1986, Bunce *et al.* 2002). However, in the 1980's it gradually became recognised that, whilst habitats had strong links to vegetation classes, they could also be independent. This was partly because animal ecologists found that vegetation structure often overrode vegetation classes but also partly because some widely recognised habitats were not directly linked to traditional vegetation associations. This situation has recently been recognised by Rodwell *et al*. (2003) showing that the match between the European Habitat Classification (EUNIS) (Davis et Moss 2002) and vegetation assemblages is often indistinct.

In the 1980's therefore, habitat mapping progressively became a separate exercise from recording vegetation alone because strategic conservation priorities did not necessarily involve the distinction between vegetation associations. For example, the small biotope project in Denmark (Agger et Brandt 1988) monitored changes in small landscape patches in intensively farmed landscapes, with minimal relationships with vegetation associations. Monitoring of changes naturally the same but differently managed landscapes using both plant communities and other patchy/linear features demonstrated indication value of landscape heterogeneity, successional contrast between neighbouring habitats, species richness and plant dispersal strategies at landscape differentiation (Kovář 1995). The Phase 1 Habitat Survey in England (JNCC 1990) enabled rapid mapping to be carried out over large areas to provide a strategic basis for determining conservation priorities albeit at a low level of detail and consistency. Similarly, an examination of the development of the Countryside Survey in the UK (Haines-Young *et al*. 2000) shows that although it initially concentrated on vegetation in 1978, by 2000 the reporting of status and change was based on 19 Broad Habitats. Whilst this project has shown the essential role of vegetation records in determining quality (i.e. favourable conservation status) it has also demonstrated that habitats are convenient for reporting. Furthermore, the habitat names are more often than vegetation units understandable by policy makers. Landscapes usually contain complexes of habitats whereas at the habitat level below contain mixtures of vegetation associations.

The list of CORINE Biotopes (Devillers *et al*. 1991) was derived from expert group discussions. It was largely based on vegetation classes and mainly concerned semi-natural habitats. Both Annex 1 of the European Habitats and Species Directive and the subsequent Palaearctic classification (Devillers et Devillers-Terschuren 1996) have strong links to the CORINE Biotope Classes. More recently, EUNIS (Davies et Moss 2002) whilst still maintaining strong links to the previous classifications, also introduced classes for artificial and highly disturbed situations. The present project originally intended to develop rules for the EUNIS classes but concluded that many terms used in the key e.g. montane and sub-Mediterranean were not sufficiently well defined for determining the classes in the field and for subsequent monitoring. Accordingly, the approach developed adopted traditional scientific principles in developing General Habitat Categories based on plant life forms, appropriate for monitoring and reporting consistently as the European scale. The present paper first describes these principles and the validation process accompanying them. The surveillance system is then described with a summary of the principal rules and the method of recording qualifiers to convey information on drivers and descriptive characters. Finally, the environmental framework for relating the necessary detailed samples to the whole population is described. Additional details on structure are also provided to provide better links of *in situ* to remote sensed information.

Principles

Surveillance and monitoring

It is useful to summarize several conceptual principles concerning basic terms relevant for the present study, such as ecological monitoring (e.g. Bunce *et al.* 2005). Surveillance is the act of survey, i.e. the recording of features at a specific location in one time frame. Monitoring involves, in contrast, repeated observation on a time-line such that change can be detected. Both approaches imply different requirements to ensure that real change is separated from observer differences.

It may be possible to survey the entire site for nature reserves (in general: for small areas), but in most cases the assessment of biodiversity or habitats must be based on samples. It is recognised that the optimum size of the sampling unit depends on the objective of study (Lambert 1972). One of the main factors deciding the characteristic of samples is that habitats often occur in patches of different sizes in contrasting landscapes. Procedures of sampling must not be compromised by spatial heterogeneity or complexity. As sampling effort is usually fixed, a choice hase made between recording many small samples units, or a smaller number of larger units. It costs more per unit area to sample many small units (as discussed by Bunce *et al.* 1996), although they may give statistically more precise estimates (Gallego 2002). Brandt *et al.* (2002a), on the other hand, argue that larger sample units provide a more systematic inclusion of variations due to management. As there is no optimal sample unit size for all the habitats and landscape at a continental scale due to variation at landscape, patch and management scales, a 1 km square is a workable compromise matching ease of survey, data content and number of sample units. Using a standard size enables the direct comparisons to be made of relative heterogeneity.

The BioHab procedure based on 1 km square sample unit is adequate to general demand that statistical inference requires samples to be drawn randomly from a defined population (e.g. Europe). In order to reduce the number of samples, a stratification should be used to partition known environmental variation and to extract stratified random samples. If this requirement is met, then the samples can be utilized to generate statistical estimates of extent for the entire defined population using standard statistical methods (Bunce *et al.* 1996). The Environmental Stratification of Europe (Metzger *et al.* 2005, Jongman *et al.* 2006) forms a suitable stratification for Europe. This dataset was derived from statistical analysis of climatic and topographic data at a 1 km square resolution. There are 84 environmental strata, which can be aggregated into 13 environmental zones.

Field habitat mapping is mostly performed as surveillance and is not intended to monitor change. Monitoring requires more stringent procedures in the same sites (Bunce *et al*. 1999) to ensure that differences recorded represent real change and not distortions due to differences between observers or recording technique, as described by Kirby *et al.* (2005). The information recorded on each occasion in a monitoring exercise has to be co-registered to identical spatial and temporal structures and employ consistent descriptive classifiers. For this purpose consistent recording rules are needed, and in the re-survey an emphasis should be placed on recognition of changes compared with the recordings made previously (e.g. the procedure used by Cooper et McCann 2002). Thus, information from the previous survey forms the basis for the field mapping and recording in the re-survey, which is implemented as a check for change of each element recorded in the previous survey. Such a repeated survey not only gives insight in environmental change, but also allows for quality control of previous surveys. Finally, it is important to note that revisiting the same sample units also has a statistical consequence. If new sample locations were selected, the total sample size would have to be increased to reatin the same statistical confidence in the estimates (Barr *et al.* 1993, Brandt *et al*. 2003). Across Europe, there is much experience in applying such methodology in the detection of change, e.g., Great Britain, Northern Ireland, Dennmark or Sweden.

Habitat definition

Long-term research and/or monitoring European habitats requires definitions that can be applied consistently in field survey across Europe (Brandt *et al.* 2002b). Habitat can be defined as: *„An element of land that can be consistently defined spatially in the field in order to define the principal environments in which organisms live.“* (Bunce *et al*. 2005). Existing European habitat classifications have been based on species, geographical location, vegetation classes or environmental factors (e.g. Davies et Moss 2002). While such attributes work well for describing habitats, they are not appropriate for standardized recording of habitats in the field.

The recording procedure presented (Bunce *et al.* 2007) adopted plant life forms, as described by Raunkiaer (1907; 1934) as the basis of the habitat categories. It is widely recognized (e.g. Walter 1973, Woodward et Rochefort 1991) that at continental level biomes need to be defined in terms of physiognomy of the dominant species and the life forms are necessary because individual species are too limited to encompass widely dispersed geographical locations. Ecological behaviour can also vary across their distribution and vicarious species also preclude the use of individual species as identifiers. A given species often shows plasticity because of environmental and local factors such as grazing or climatic stress (e.g. dominant tree at the ecocline between forest and subalpine belt). Further advantages of using life forms are that they provide direct links between *in situ* data and dynamic global vegetation models (e.g. Sitch *et al.* 2003), but also with earth observation data because of their correspondence vegetation structure.

Originally plant life forms were identified as valuable in the project because they provided rules to separate grassland, scrub and forest categories using rules that could be applied consistently in the field. However, during the project it became clear that life forms provided a means of transcending species and enabling consistent recording of habitats to beundertaken. It was also realised in the project that the adoption of life forms would provide links between European categories and other studies of global change that use biomes based largely on life forms e.g. Mediterranean scrub in the western USA, Chile, South Africa and Australia. The basis of the General Habitat Categories GHC'- classification of plant life forms produced by Raunkiaer (1934) - can also be sufficiently robust to be used to link existing datasets which have been collected for monitoring. The underlying scientific hypothesis is that habitat structure is related to the environment on the European scale, or even locally if there is a sufficiently wide range of conditions.

The application of the Environmental Stratification (Jongman *et al.* 2006) mentioned above provides a sampling framework linked to climate, topography and geographical location, which can also be tested statistically. Various floras were consulted, especially Clapham *et al.* (1952) and Pignatti (1982) to determine at what level to treat life forms as some recent floras e.g. Oberdorfer *et al.* (1990) give highly detailed categories. However, as Raunkiaer (1934) originally emphasized, the more detailed breakdown of life forms, loses the strong relationship with climate. Eventually, it was decided to use 16 Life Forms

(Herbaceous and Tree/scrub, see Table 1) with the plant height ranges taken from more recent literature - e.g. di Castri *et al*. (1981); Quezel et Barbero (1982). The main problem was however, with *Gramineae, Cyperaceae* and *Juncaceae*, where many species have rhizomes, which are primarily for vegetational reproduction not for perennation. There are also differences between the attribution floras of life forms as well as difficulties in the determination of the actual position of the rhizomes or stolons in the field. It was therefore decided to group these three taxa together as "caespitose hermicryptophytes". Further details and examples of the species in the 16 life forms are given in Bunce *et al*. (2005). It is also recognised that some species are sufficiently plastic to adapt to several habitats, e.g. *Ranunculus aquatilis*; in which case the environmental conditions present at the site, as described below should be used to determine whether, for example, it is in aquatic or waterlogged conditions.

Another aspect of plasticity relates to woody species which respond to a range of environmental and management pressures. These species can occur in lower than optimum height categories because:

- They have been heavily grazed
- They have been burnt
- They are regenerating
- They are in highly exposed or extreme environments
- They are degraded (in various degree) by the artificial substitution of dominant species

The first three categories are transitional and shifts can take place according to changes in external pressures e.g. fire or felling. The fourth is a climax state. The fifth represents human ways of the stand cultivation manifested by geometric design in seedling pattern, their age homogeneity, substitution of natural trees with alochtonous (e.g., deciduous ones by conifers, which may eliminate herb species diversity through shading, allelopathy etc.). The only way to provide consistent data is to record the actual heights of the tree and shrub cover in the field, because otherwise the potential height is a matter of judgement. There is a functional relationship between the life forms and pathways of change over time e.g. from tall shrubs 2-5 m to forest trees over 5 m which are then quantifiable, as in the flow diagrams between habitats given by Haines-Young *et al.* (2000).

Land associated with built structures and routes of communication (termed „Urban" in a broad sense) and agricultural cropland (termed „Crops") cannot be defined solely in terms of life forms as they are primarily land uses. However, for policy and practical reasons it is essential that such land is separated from other land covers that are mainly in agricultural or forest use. Hence, these two categories have been separated as „super categories" at the first level of the hierarchy (Bunce *et al.* 2005), together with bare land as shown in Table 1. However, within both the former categories („Urban" and „Crops"), subsequent divisions are then based on lifeforms at the second level of Table 1. In addition, the „Sparsely Vegetated" super category is separated to cover land with vegetation cover below 30 %, e.g. glaciers. The other super categories are based on life forms.

A major problem of theoretical habitat classifications for field recording is the proliferation of classes. In some habitat classifications by e.g. Morillo Fernandez (2003) there are almost 1000 classes and in EUNIS there are 350 at level three. It was therefore decided that below the first tier of five super categories all possible combinations of life forms should be included, even although some will be rare (Table 1). This procedure has provided a statistical rule for determining the number of General Habitats Categories (GHC's) and results in 130 covering the pan-European region, except Turkey.

Table 1. Super categories of Habitat types and the related General Habitats categories (Bunce *et al*. 2005).

Urban (URB)	Artificial (ART)	
	Non-vegetated (NON)	
	Vegetables (VEG)	
	Herbaceous (GRA)	
	Woody (TRE)	
	Combinations	
Crops (CUL)	Cultivated bare ground (SPA)	
	Cultivated herbaceous crops (CRO)	
	Woody crops (WOC)	
	Combinations	
Sparsely Vegetated (SPV)	Sea (SEA)	
	Marine (MAR)	
	Aquatic (AQU)	
	Terrestrial (TER)	
	Ice and snow (ICE)	
	Combinations	
Vegetated Herbaceous (HER)	Submerged hydrophytes (SHY)	
	Emergent hydrophytes (EHY)	
	Helophytes (HEL)	
	Leafy hemicryptophytes (LHE)	
	Caespitose hemicryptophytes (CHE)	
	Therophytes (THE)	
	Succulents (SUC)	
	Geophytes (GEO)	
	Herbaceous chamaephytes (HCH)	
	Cryptogams (CRY)	
	Combinations	
Vegetated tree/shrub (TRS)	Dwarf chamaephytes (< 0.05 m) (DCH)	Winter deciduous (DEC)
	Shrubby chamaephytes (0.05-0.30 m) (SCH)	Evergreen (EVR)
	Low phanerophytes (0.30-0.6 m) (LPH)	Coniferous (CON)
	Mid phanerophytes (0.6 – 2 m) (MPH)	Non-leafy evergreen (NLE)
	Tall phanerophytes (2- 5 m) (TPH)	Summer deciduous and/or spiny cushion (SPI)
	Forest phanerophytes (>5 m) (FPH)	Combinations

Other life forms, e.g. tall succulents would have to be included for other continents, but at present they are included as qualifiers. This restricted list acts as a lowest common denominator and enables the primary decision to be made in the field, or to be derived from extant data (e.g. phytosociological relevés). A worked example of monitoring national change using a similar level of categories of habitats is given by Haines-Young *et al.* (2000).

The determination of the GHC is based upon a series of five dichotomous divisions related to the six. These determine the set of life forms that can be used to identify the appropriate GHC. The first decision concerns whether the element is „Urban“, the second whether it is a „Crop“, the third whether is „Sparsely Vegetated“, the fourth whether it is „Trees or Shrubs“, and the fifth whether it is „Wetland“. As discussed, rules have added for further divisions in all super categories and habitat categories, including percentage criteria. New areal and linear elements are separated adjacent or surrounding areal elements based on pre-defined rules as described by Bunce *et al.* (2005).

Additional qualifiers

Specific additional qualifiers are constituted for further description of the GHCs and essential for determination of landscape ecological parameters. We can establish lists of qualifiers: global (e.g. percentage cover), environmental (e.g. soil moisture), site (e.g. moraine), management (e.g. cattle grazing, meadow mowing or cultural forest plantation). These qualifiers are recorded in combination with the GHC to provide information on variation between elements that may have the same GHC. A matrix of environmental conditions was constructed, as described by Bunce *et al.* (2005), with moisture classes on the horizontal axis. Moisture classes suitable for application across the range of European habitats were adapted from Pyatt (1999). The soil factors are based on indicator values originally developed by Ellenberg *et al.* (1992) for Central Europe using expert knowledge of the environmental amplitude of individual species. The species have been recalibrated for Britain (Hill *et al*. 2000). Hovever, they are not available for many regions, so local experience of the ecological amplitude of indicator species is often needed. For all cells in the matrix the overall balance of species should be used, not individual indicator plants (e.g. in the Pannonian region the presence of some individuals of *Melica ciliata* is insufficient to assign the term „xeric“ to the element).

The recording procedure

Preparation

It is emphasized (Bunce *et al.* 2007) that no continent-wide survey can be carried out without adequate field training for all surveyors to ensure that terms are fully understood and interpreted in the same way. For example, environmental terms are often used within a local context, e.g. „dry“ in Scotland may be „mesic“ compared with southern Italy. Surveyors across Europe therefore need to be familiar with predefined environmental categories. In the field, combined teams of two people, probably consisting of a botanist and an experienced cartographer, are needed to ensure that required expertise is available.

For survey, the recording of the GHCs should be based on the overall phenology of the region (Bunce *et al.* 2007). This means that in the Mediterranean region the recording period will be earlier than in cetral and northern Europe. Planning the recording dates of the field survey needs to take into account differences in seasons between years, so some flexibility is needed. Repeat visits for monitoring should then be carried out as close as possible to the data of the original visit, assuming that there is no shift in timings of the season (Barr *et al.* 1993) because it has been shown that differences between dates of survey is a major source of noise in change statistics. The extent of the window needs to be set by region, using local phenological information and differs between environmental zones, environmental strata and countries.

Data quality control (i.e. supervision of surveyors) and assurance (i.e. independent checks of recording) are all essential to produce robust data. Barr *et al.* (1993) analysed random checks of comparable categories to GHCs and showed a correspondence of 84 %. Any future program would need to incorporate such checks, so that European policy makers would have confidence in the results.

All major decisions should be made in the field (Bunce *et al.* 2007). It is recognized that GHCs represent simplification, but this is necessary to obtain consistency across the continent. However, further details of life forms present and dominant species are also included in the recording procedure. It is preferable to carry out preparatory work on delineation of the major elements within the survey are from aerial photographs and related material, e.g. cadastral maps, preferably at a 1:10,000 scale, to assist the mapping process. The surveyor can then go in to the field with a base map with parcel outlines. While some borders may need to be adjusted or added, such preparatory work reduces time spent in the field. At a later stage, it is also possible to extract other data in the laboratory from extant datasets (e.g. slope angles, and aspect).

Areal elements

The procedure was initially developed for mapping 1 km square samples, but is also suitable for other scales, e.g. Cooper et McCann (2002) used 0.5 x 0.5 km squares and Bloch-Petersen *et al.* (2006) applied the GHCs to small biotopes. Within the 1 km^2 sample unit, the surveyor delineates all habitats with an area greater than 400 m^2 (Minimal Mappable Element – MME). For each delineated unit the surveyor determines the GHC (Field 1) and subsequent environment, site and management qualifiers, which are listed in sequential fields on the recording sheet (Fields 2-4). Next, the surveyor records all life forms with a cover of over 10 %, and all individual species or crops with a vegetation cover pf over 30 % to the mapping unit (Field 5). Three further fields are provided for the pan-European habitat classifications (e.g. EUNIS), local habitat classifications (e.g. Morillo Fernandez 2003) and for phytosociological associations (e.g. Rodwell *et al.* 2002).

Although the MME has to occupy at least 400 m^2 it can be a complete shape, so long as the shortest measurement is 5 m, based on field trials in 1998 and 1999 in El Tiemblo near Madrid, Spain. This contrasts with the 10,000 m^2 (100 m x 100 m) of the CORINE land cover is essential to express the landscape ecological characteristics of small scale landscapes, e.g. Crete or Brittany.

Bunce *et al.* (2005) provide detailed rules for mapping, including rules for consistent mapping of gradients, element that cross the boundaries of the 1 km sample square, and the use of interpreted aerial photograph to provide initial boundaries for mapping. Some elements, e.g. motorways will be mapped areal elements, but may be subsequently allocated to linear features by database management for specific objectives (e.g. Haines-Young *et al.* 2000). The fundamental principle is that disaggregated data are collected, so that subsequent analysis can produce statistics that are sufficiently flexible to answer a range of policy requirements.

Linear and point elements

In the majority of literature cited above both linear and point features are largely omitted. This also applies to most phytosociological studies e.g. in the Czech Republic over 30,000 relevés have been recorded, none smaller part of which is detectable are as on linear or point features. However, many studies have shown that especially in intensively managed agricultural landscapes, biodiversity has progressively become restricted to such situations e.g. Bunce et Hallam (1991), Kovář (1997) and Hermy et de Blust (1997). Moreover, as intensification continues, so does the pressure increase even in such limited areas e.g. Haines-Young *et al*. (2000) and Agger et Brandt (1988). Furthermore, many cultural landscapes are exceptionally rich in linear features largely the product of management by managing the terraced landscapes of Crete and the dense hedgerow network of the bocage. It is therefore essential not only to assess the resources of linear and point elements in representative landscapes but also to monitor change. As with areal elements a series of rules and protocols have therefore been developed to maintain consistency and repeatability as described by Bunce *et al.* (2005).

Linear elements have a Minimal Mappable Length (MML) of 30 m. Those features that comprise only of vegetation must be wider than 0.5 m in order to exclude narrow strips (e.g. lines of vegetation beside walls). Elements that are smaller than 400 m^2 and shorter than 30 m can be recorded as points. Linear habitats often occur as complexes, e.g. a fence, a ditch and a hedge, in which case instructions are provided for mapping so that a given combination is always recorded by single alpha-numeric code incorporating its detailed composition.

It is recognised that in many cultural landscapes the number of point features can be very large, e.g trees in parkland or patches of rocks in fields. Two guidelines are provided for recording such points. Firstly, the recorded point features should add to landscape diversity, usually because they represent a particular habitat which is generally absent from the surrounding area, e.g. rock outcrops or boulders in a grass field. Secondly, the recorded point features should also have an effect on the ecological functioning of landscapes, e.g. small water bodies which act as drinking places in grasslands or weirs in watercourses which hinder migration of fish. However, a given survey may decide to omit point features, in which case the procedure followed should be documented, or they may be recorded using a global qualifier such as „scattered“.

Management qualifiers reflecting plant cover syngenesis: degree of naturalness

Variability of managament practices will need more detailed system of qualifying the particular influences on both patchy and linear natural or (semi)cultural components in landscape. Significant differences in management indication and future needs are clear in e.g. forests (tree floor is artificially changed or not), herb cultures (weeds are eliminated or not), hedgerows (they are directed as functional barriers against pollution and erosion) or plant assemblages on industrial deposits (assisted vegetation succession is supported in according to substrate toxicity) – e.g. Kovář *et al.* 1997, Kovář 2004a.

In the context of additional habitat qualifying, at level of local territory a list of phytosociological units (associations, in some cases: alliances) becomes suitable for balanced indication of the habitat state (habitat maturity, habitat naturalness, habitat genesis). This is also applicable for the Czech Republic and, generally, for Central Europe, where classical phytosociology was traditionally developed (Braun-Blanquet 1928; Tüxen 1937). It offers good and relatively detailed knowledge on potential natural conditions nearly all of the area (Neuhäuslová *et al.* 1998). Examples of derived products directed to nature conservation and habitat classification for land use planning are e.g. Moravec *et al.* 1995 or Chytrý *et al*. 2001. While highly formalized approach (using e.g. COCKTAIL method for classification of communities) is superior for large-scale vegetation surveys (Bruelheide et Jandt 1997, Chytrý 2000) it looses indicative values because most of the community character-species exhibit serious variations in ecological behaviour continentally. Imperfectly formalized approaches typical for studies of smaller areas (e.g. river basin, mountain range or political district) can be compensated by very good field knowledge of vegetation variability and the relationships between basic vegetation units and environmental properties by their authors. Basic vegetation units resulted from classical procedure in the relevant frame of 1 km^2 territory could serve as good site qualifier (plant life forms suitable for GHC identification across the scale of continents, e.g. Europe, are not enough to distinguish fine-scale habitat features which can be highly teritorially specific). In semicultural or cultural plant assemblages, such as forests with planted trees or meadows with sown dominants, we can use phytosociological indices of similarity (syngenetic approach to naturalness) as management qualifiers to be consistent in the method applied to this level (Hadač et Sofron 1980). The application of this method in the context of surveillance and monitoring habitats has not published (it was firstly presented orally in the BioHab Prague workshop (Kovář 2004b).

From the viewpoint of macroclimatic conditions, the Czech Republic represents a forested territory typical of Central Europe (open vegetation formations such as alpine meadows, extrazonal steppes or peatlands are of negligible size). Map of potential natural vegetation of this area (Neuhäuslová et al. 1998) shows highly structured mosaics of the original vegetation cover. High diversity of ecosystems is significantly influenced by diversified chemistry and physical properties of the geological substrate (sedimentary and volcanic rocks, basiphilous and acidic material), diversified relief in landforms (in the altitudes from 200 to 1600 m), with a dense river network on the hydrological „roof of Europe“, and climatic features (from suboceanic to semi-continental character).

Neolithic impact of mankind is dated approximately to 6500 B.P. and consists in deforestation, cultivation of crop plants and forest grazing, in modern period also in environmental pollution and expansion of allien species introduced both artificially (*Quercus rubra, Pseudotsuga menziesii, Abies grandis, Pinus nigra* etc.) and spontaneously (*Robinia pseudoacacia, Ailanthus altissima, Acer negundo* etc.).Very roughly, actual forested cover of the CR occupies approx. one third of the whole area and maybe 80 % of woody stands are formed by cultures. This results from the last 150 years of forestry practice of substituting the autochtonous (mainly broad-leaved) trees with Norway Spruce and Scots Pine which caused such impacts as fragmentation of forest complexes due to windstorms or pest population bursts.

Our field experience is that we can meet forests of similar plant composition but with different history (syngenesis). The problem of interpretation is how to express different syngenesis of forests when high phytocoenological identity and/or similarity is achieved. The authors Hadač et Sofron (1980) declared that cultural forests can form true plant communities which are part of the system of natural plant communities. As the differences between the cultural and natural forests may be on various hierarchical levels (in the sense of classical European phytosociology: facies, variant, subassociation, association, alliance…) and of a different degree, it is desirable to take this fact into account.

There are three possibilities:

1. climax community (like *Quercus petraea* in *Potentillo-Quercetum* or *Picea excelsa* in *Calamagrostio villosae-Piceetum*),

2. planted trees are of the same species as the original species, or locally planted trees belong to other species, but with similar character, e.g. mesotrophic deciduous tree species, planted instead of other deciduous species, or conifers instead of other conifer species,

3. planted trees belong to species of quite different ecological character (like *Picea excelsa* or *Robinia pseudacacia* versus *Carpinus betulus* or *Quercus robur*).

Point 1 means only a few or no changes in the composition of the shrub- and herb layer, and such communities are usually incorporated in the „natural“ system of plant communities (it corresponds with relatively low ecosystem lability). Second and third cases deviate gradually more and more from the natural one with decreasing degree of ecological stability. Hadač et Sofron (1980) suggested to use Jaccard index of similarity as a formal descriptor of the relative naturalness:

$$Q_j = \frac{c}{a + b - c} \times 100$$

a - number of species in relevé 1
b - number of species in relevé 2
c - number of common species

Vegetation ecologists know that communities belonging to the same association have usually a Jaccard index higher than 45 (often over 50). Stands belonging to different associations but to the same alliance, usually have an index of 20-35, and communities with a lower index belong usually to different orders.

We can thus compare the studied forest communities with the nearest similar natural community in spite of the fact that individual trees grow in regular arrangement of rows and lines (they are planted out). If we know that our landscape types with their plant species diversity and plant species abundance correspond more or less with the phytocenological level of alliance and/or group of associations, our position is in the middle level of Jaccard index values. Hence we have simple quantitative descriptor for the assessment of the forest naturalness.

In other words, it is useful to apply this parameter as another important management qualifier (**qualifier of syngenesis**) within recording sheets for the forest land cover, e.g., in the following way:

degree of naturalness	Jaccard index
1	< 45
2	20 - 45
3	> 20

To distinguish in practice between structurally similar but syngenetically different ecosystems, Hadač et Sofron (1980) suggested a nomenclatoric solution - the prefix „culti“ in the latin name of a phytocenological unit (with planted dominant), e.g. *Vaccinio myrtilli-*

culti-Piceetum (analogically: *-culti-Quercetum, -culti-Alnetum*, etc.). It is our job now, to transfer this into the formal nomenclature of habitat (landscape) classification.

Suggestion for evaluating:

degree of naturalness		nomenclature	abbreviation
1	high	cultural habitat (landscape) of high naturalness	CN
2	medium	cultural habitat (landscape) of semi-naturalness	CS
3	low	cultural habitat (landscape) of low naturalness	CL

The rare effectively protected or untouched forests (more frequent e.g. in the Ukrainian Carpathians) which could be declared as really (syngenetically) natural ones, might be called natural forests (N).

The same principles could be used for the other habitats systematically influenced and/or managed by humans, e.g. meadows with sown of preferred species.

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References

AGGER, P., BRANDT, J., 1988: Dynamics of small biotopes in Danish agricultural landscapes. Landscape Ecology, 1: 227-240.

Barr, C.J., Bunce, R.G.H., Clarke, R.T., Fuller, R.M., Furse, M.T., Gillespie, M.K., Groom, G.B., Hallam, C.J., Hornung, M., Howard, D.C., Ness, M.J., 1993: COUNTRYSIDE SURVEY 1990: MAIN REPORT VOLUME 2. DEPARTMENT OF THE ENVIRONMENT, LONDON, 174 P.

Bloch-Petersen, M., Brandt, J., Olsen, M., 2006: INTEGRATION OF HABITAT MONITORING BASE ON PLANT LIFE FORM COMPOSITION AS AN INDICATOR OF ENVIRONMENTAL CHANGE AND CHANGE IN BIODIVERSITY. DANISH J.GEOGR., 106(2): 41-54.

BRANDT, J., BUNCE, R.G.H., HOWARD, D.C., PETIT, S., 2002a: General principles of monitoring land cover change based on two case studies in Britain and Denmark. Landscape and Urban Planning, 62: 37-51.

BRANDT, J., HOLMES, E., RAVN, H.P., 2002b: Biodiversity and trees outside forests: The case of Denmark. In: PUUMALAINEN, J., KENNEDY, P., FOLVING, S. (eds.): Forest Biodiversity – Assessment Approaches for Europe, EUR Report 20423 EN. Joint Research Centre (DG JRC). Ispra European Communities, pp. 80-85.

BRANDT, J., DE BLUST, G., WASCHER, D., 2003: Monitoring multifunctional terrestrial landscapes. In: BRANDT, J., VEJRE, H. (eds.): Multifunctional landscapes. Vol. II: Monitoring, Diversity and Management. WITPress, Southampton, pp. 75-84.

BRAUN-BLANQUET, J., 1928: Pflanzensoziologie. Grundzuge die Vegetationskunde. Springer verlag, Berlin, 330 p.

BRUELHEIDE, H., JANDT, U., 1997: Demarcation of communities in large databases. – Phytocoenologia, 27: 141 – 159.

BUNCE, R.G.H., SHAW, M.W., 1973: A standardised procedure for ecological survey. Journal of Environmental Management, 1: 239-258.

Bunce, R.G.H., Hallam, C.J., 1991: Diversity of vegetation in British landscapes. In: Pineda, F.D., Casado, M.A., de Miguel, J.M., Montalvo, J. (eds.), Biological Diversity. Fundacion Ramon Aceres, Madrid, pp 77-81.

Bunce, R.G.H., Howard, D.C. Hallam, C.J., Barr, C.J., Benefield, C.B., 1993: Ecological consequences of land use change. Countryside Survey 1990, Volume 1. Department of the Environment, London, 129 p.

Bunce, R.G.H., Barr, C.J., Clarke, R.T., Howard, D.C., Lane, A.M.J., 1996: Land classification for strategic ecological survey. Journal of Environmental Managmeent, 47: 37-60.

Bunce, R.G.H., Smart, S.M., van de Poll, H.M., Watkins, J.W., Scott, W.A., 1999: Measuring Change in British Vegetation. ECOFACT Volume 2. NERC-CEH, pp. 144.

Bunce, R.G.H., Carey, P.D., Elena-Rossello, R., Orr, J., Watkins, J.W., Fuller, R., 2002: A comparison of different biogeographical classifications of Europe, Great Britain and Spain. Journal of Environmental Management, 65: 121-134.

Bunce, R.G.H., Groom, G.B., Jongman R.G.H., Padoa-Schioppa, E. (eds.), 2005: Handbook of Surveillance and Monitoring of European Habitats. Alterra, Wageningen, 107

Bunce, R.G.H., Metzger, M.J., Brandt, J., de Blust, G., Elena-Rosello, R., Groom G., Halada, L., Hofer, G., Howard, D.C., Jongman R.G.H., Kovář, P., Műcher, C.A., Padoa-Schioppa, E, Paelinckx, D., Palo, A., Perez-Soba, M., Ramos, I., Roche, P., Skanes, H., Wrbka, T., 2007*: A standardized procedure for surveillance and monitoring of European habitats. Landscape Ecology (in press).*

Cawsey, E.M., Austin, M.P., Baker, B.L., 2002: Regional vegetation mapping in Australia: a case study in the practical use of statistical modelling. Biodiversity and Conservation, 11: 2239-2274.

Chytrý, M., 2000: Formalized approaches to phytosociological vegetation classification. - Preslia, 72: 1-29. [In Czech]

Chytrý, M., Kučera, T., Kočí, M., (eds.), 2001: Katalog biotopů České republiky. Habitat Catalogue of the Czech Republic. – Agentura ochrany přírody a krajiny ČR, Praha [in Czech].

Clapham, A.R., Tutin, T.G., Warburg, E.F., 1952: Flora of the British Isles. Cambridge University Press, 1591 p.

Cooper, A., McCann, T., 2002: Habitat change in the Northern Ireland Countryside: summary report of the Northern Ireland Countryside Survey 2000. Environment and Heritage Service, Department of the Environment for Northern Ireland, Belfast.

Davies, C.E., Moss, D., 2002: EUNIS Habitat Classification. Final Report to the European Topic Centre of Nature Protection and Biodiversity, European Environment Agency, 125 p.

den Boer, P.J., van Dijk, T.S., 1994: Carabid beetles in a changing environment. Wageningen Agricultural University Papers, 30 p.

Devillers, P., Devillers-Terschuren, J., Ledant, J.P., 1991: CORINE biotopes manual. Habitats of the European Community. Data Specifications – Part 2. Office for Official Publications of the Euroepan Community, Luxembourg, 300 p.

Devillers, P., Devillers-Terschuren, J., 1996: A classification of palearctic habitats and preliminary habitats in Council of Europe Member States. Report to the Council of Europe Convention on the Conservation of European Wildlife and Natural Habitats, 268 p.

Di Castri, F., Goodall, D.W., Specht, R.L., (eds.), 1981: Mediterranean-type ecosystems of the world. Elsevier, Amsterdam, 52 p.

ELENA-ROSSELLO, R., (ed.), 1997: Classification Biogeoclimatica de Espaňa Peninsula y Baleaur. M.A.P.A., Madrid, 448 p.

ELLENBERG, H., WEBER, H.E., DULL, R., WIRTH, V., WERNER, W., PAULIBEN, D., 1992: "Zeigerwerte von Pflanzen in Mitteleuropea". Scripta Geobotanica 18 (2nd edition).

EUROPEAN COMMUNITIES, 1992: COUNCIL DIRECTIVE 92/43/EEC ON THE CONSERVATION OF NATURAL HABITATS AND OF WILD FAUNA AND FLORA. TEXT AND ANNEXES OF THE HABITAT DIRECTIVE. OFFICE FOR OFFICIAL PUBLICATIONS OF THE EUROPEAN COMMUNITIES, 21 MAY 1992.

FJELLSTAD, W.J., DRAMSTAD, W.E., STRAND, G.H., FRY, G.L., 2001: Heterogeneity as a measure of spatial pattern for monitoring agricultural landscapes. Norwegian Journal of Geography, 55: 71-76.

Gallego, J., 2002: BUILDING AGRO ENVIRONMENT INDICATORS. FOCUSSING ON THE EUROPEAN AREA FRAME SURVEY LUCAS. EUR REPORT 20521N. EUROPEAN COMMISSION, ISPRA.

GRABHERR, R., GOTTFRIED, M., PAULI, H., 1994: Climate effects on mountain plants. Nature, 369: 448-448.

GREIG-SMITH, P., 1964: Quantitative plant ecology. 2nd edition. Butterworth, London, 256 p.

HADAČ, E., SOFRON, J., 1980: Notes on syntaxonomy of cultural forest communities. – Folia Geobotanica et Phytotaxonomica, 15: 245-258.

HAINES-YOUNG, R.H., BARR, C.J., BLACK, H.I.J., BRIGGS, D.J., BUNCE, R.G.H., CLARKE, R.T., COOPER, A., DAWSON, F.H., FIRBANK, L.G., FULLER, R.M., FUIRSE, M.T., GILLESPIE, M.K., HILL, R., HORNUNG, M., HOWARD, D.C., MCCANN, T., MORECROFT, M.D., PETIT, S., SIER, A.R.J., SMART, S.M., SMITH, G.M., STOTT, A.P., STUART, R.C., WATKINS, J.W., 2000:. Accounting for nature: assessing habitats in the UK countryside. DETER, London, 134 p.

HERMY, M., DE BLUST, G., 1997: Punten en Lijnen in het Landschap. Schuyt en Co, Haarlem, 335 p.

HILL, M.O., ROY, D.B., MOUNTFORD, J.O., BUNCE, R.G.H., 2000: Extending Ellenberg's indicator values to a new area: an algorithmic approach. Journal of Applied Ecology, 37: 3-15.

Intergovernmental Group on Earth Observation. 2004: THE GLOBAL EARTH OBSERVATION SYSTEM OF SYSTEMS (GEOSS). 10-YEAR IMPLEMENTATION PLAN.

JNCC, 1990: Handbook for Phase 1 Habitat Survey: a technique for environmental audit. 2 volumes. Joint Nature Conservancy Council, Peterborough, Vols 1 & 2, 143 p.

JONGMAN, R.H.G., BUNCE, R.G.H., METZGER, M.J., MŰCHER, C.A., HOWARD, D.C., 2006: A statistical Environmental Classification of Europe: objectives and applications. Landscape Ecology, 21: 409-419.

KIRBY, K.J., SMART, S.M., BLACK, H.I.J., BUNCE, R.G.H., CORNEY, P.M., SMITHERS, R.J., 2005: Long term ecological change in British woodland (1971-2001). A resurvey and analysis of change based on the 103 sites in the Nature Conservancy 'Bunce 1971' woodland survey. English Nature Resarch Report No 653, English Nature, Peterborough, 139 p.

KOVÁŘ, P., 1995: Is plant community organization level relevant to monitoring landscape heterogeneity? Landscape and Urban Planning, 32: 137-151.

KOVÁŘ, P., 1997: Biodiversity of hedgerows surrounding a point source of nitrogen pollution (Wensleydale, Northern England). Thaiszia-J.Bot., 7: 65-73.

KOVÁŘ, P., 2004a: Abandoned anthropogenic landscapes: Are they potentially multifunctional? In: Kovář, P. (ed.), Natural Recovery of Human-Made Deposits in Landscape (Biotic Interactions and Ore/Ash-Slag Artificial Ecosystems). Academia, Prague, pp. 352-358.

KOVÁŘ, P., 2004b: Syngenetic approach to classification of (semi)cultural forests. – Presented in BioHab workshop "Linking regional vegetation classification to European habitats", Prague, April 26-28, 2004.

KOVÁŘ, P., LEPŠ, J., 1986: Ruderal communities of the railway station Česká Třebová (Eastern Bohemia, Czechoslovakia) - remarks on the application of classical and numerical methods of classification. Preslia, 58: 151-163.

KOVÁŘ, P., KOVÁŘOVÁ, M., BUNCE, R., INESON, P., BRABEC, E., 1997: Role of hedgerows as nitrogen sink in agricultural landscape of Wensleydale, Northern England. Preslia 68, (3/1996): 273-284, 1997.

LAMBERT, J.M., 1972: Theoretical models for large-scale vegetation survey. In: Jeffers, J.N.R., (ed.), Mathematical models in ecology. Blackwell, Oxford, pp. 87-109.

METZGER, M.J., BUNCE, R.G.H., JONGMAN, R.H.G., MŰCHER, C.A., WATKINS, J., 2005: A climatic stratificiation of the environment of Europe. Global Ecology and Biogeogrtaphy, 14: 549-563.

MORAVEC, J. et al. 1995: Rostlinná společenstva České republiky a jejich ohrožení. Red list of plant communities of the Czech Republic and their endangerment. Ed. 2. – Severočes. Přír., suppl. 1995: 1 – 206.

MORILLO FERNANDEZ, C., 2003: Manual de Interpretación de los Hábitats de Espaňa, Vols. 1 & 2. TRAGSA, Ministerio de Medio Ambiente, Madrid, 479 p.

MŰcher, C.A., Hennekens, S.M., Bunce, R.G.H., Schaminee, J.H.J., 2004: MAPPING EUROPEAN HABITATS TO SUPPORT THE DESIGN AND IMPLEMENTATION OF A PAN-EUROPEAN NETWORK, ALTERRA 952, ALTERRA, WAGENINGEN.

NEUHÄUSLOVÁ, Z. et al. 1998: Map of potential natural vegetation of the Czech Republic. – Academia, Prague.

OBERDORFER, E., MUELLER, T., KORNECK, D., LIPPERT, W., MAKGRAF-DANNENBERG, I., 1990: Pflanzensoziologische Exkursionsflora. 6th edition. Stuttgart Ulmer, 1050 p.

PETIT, S., FIRBANK, L., WYATT, B., HOWARD, D., 2001: MIRABEL. Models for integrated assessment of biodiversity in European landscapes, AMBIO, 30: 81-88.

PIGNATTI, S., 1982: La flora d'Italia. Edagricole, Bologna, 2324 p.

PYATT, D.G., 1998: Correlation of European Forest Site Classifications. EU Concerted Action Final Report, Forest Reseach Northern Station, Roslin.

QUEZEL, P., BARBERO, M., 1982: Definition and characterization of Mediterranean-type ecosystems. Ecologia Mediterranea, 7: 15-29.

RAUNKIAER, C., 1907: Planterigets livsformer og deres Betydning for Geografien. Munksgaard, Copenhagen, 102 p.

RAUNKIAER, C., 1934: The Life Forms of plants and staitsical plant geography, being the collected papers of C. Raunkiaer. Clarendon, Oxford, 632 p.

RODWELL, J.S., SCHAMINEE, J.H.J., MUCINA, L., PIGNATTI, S., DRING, J., MOSS, D., 2002: The Diversity of European Vegetation. An overview of phytosociological alliances and their relationships to EUNIS habitats. Ministry of Agriculture Nature Management and Fisheries, The Netherlands and European Environment Agency, Wageningen, 168 p.

SHEAIL, J., BUNCE, R.G.H., 2003: The development and scientific principles of an environmental classification for strategic ecological survey in Great Britain. Environmental Conservation, 30: 147-159.

SITCH, S., SMITH, B., PRENTICE, I.C. ET AL. 2003: Evaluation of ecosystem dynamics, plant geography and terestrial carbon cycling in the LPJ Dynamic Vegetation Model. Global Change Biolog, 9: 161-185.

TÜXEN, R., 1937: Die Pflanzengesellschaften Nordwestdeutschlands. Mitteilungen der Floristisch-soziologischen Arbeitsgemeinschaft in Niedersachsen 3: 1-170.

Walter, H., 1973: VEGETATION OF THE EARTH IN RELATION TO CLIMATE AND ECO-PHYSIOLOGICAL CONDITIONS. SPRINGER VERLAG, NEW YORK, 237 P.

WOODWARD, F.I., ROCHEFORT, L., 1991: Sensitivity analysis of vegetation diversity to environmental change. Global Ecology and Biogeography Letters, 1: 7-23.