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NatureServe Technical Report ▪ July 2009

Biodiversity Inventory of Natural Lands

A How-To Manual for Foresters and Biologists

APPENDICES

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July 2009

Citation: Cutko, A. 2009. *Biodiversity Inventory of Natural Lands: A How-To Manual for Foresters and Biologists*. Arlington, Virginia: NatureServe.

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Species Inventory Manuals Available from the British Columbia Conservation Data Centre

To access these manuals, see http://www.env.gov.bc.ca/wildlife/wsi/manuals.htm.

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Appendix B

Overview of Ecological Classification

The International Vegetation Classification system (IVC) is used by most natural heritage programs and federal agencies in the United States. (Many natural heritage programs in the Eastern U.S. use classification systems uniquely developed for their own states. These classifications are linked or cross-walked to the IVC). With support from The Nature Conservancy and in collaboration of the Ecological Society of America, U.S. Geologic Survey, and U.S. Federal Geographic Data Committee, Grossman et al (1998) produced the first comprehensive draft of the US-NVC, based in part on modifications to a United Nations Educational, Scientific, and Cultural Organization international vegetation classification and mapping system (UNESCO 1973). At about the same time, the Federal Geographic Data Committee (1997) adopted a slightly modified version of this classification system as a federal standard for all agencies.

Work in Canada is underway along similar conceptual lines, using the framework of the IVC to guide development of vegetation types (Ponomarenko and Alvo 2000; Alvo and Ponomarenko 2003). IVC partners in Canada are working to ensure that the classification will serve provincial, national, and international needs. Like the US-NVC, the CNVC is building on the classification work done by provincial or local ecologists. Many provinces have already developed provincial or sub-provincial Forest Ecosystem Classifications (FECs). The Canadian Forest Service is currently working closely with provincial governments and conservation data centers to link provincial forest and woodland types with any defined associations of the CNVC.

The overall IVC classification framework has multiple hierarchical levels that allow it to be applied at the spatial level appropriate to a range of conservation and management activities. Five levels (formation class, formation subclass, formation group, formation subgroup, and **formation**) are based on vegetative structure or physiognomy, and the two finer levels (alliance and association) are derived from species composition (floristics). Only the finest level of classification—the association—receives a conservation status assessment (i.e., Global and State rarity rank), and for the past decade it has been the most frequently used level for conservation purposes, including forest certification. NatureServe follows the definition of association provided by Jennings et al (2003) as "a vegetation classification unit defined on the basis of a characteristic range of species composition, diagnostic species occurrence, habitat conditions, and physiognomy."

In the U.S and Latin America, the Ecological Systems classification is a relatively new, mid-scale classification that describes landscapes in terms of their component US-NVC alliances and associations. Ecological systems represent recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding. NatureServe's North American systems classification describes over 600 upland and wetland system types found in the lower 48 United States, southern Alaska, and adjacent portions of Mexico and Canada. The nesting of associations within systems allows users to go back and forth between the two classification approaches. An ecological systems classification is also available for Latin America, where nearly 700 ecological systems have been described (Josse et al. 2003). In many regards the ecological system classifications provide a more effective conservation planning tool for broader regions.

As noted previously, most states in the eastern U.S. have their own classifications, except for Tennessee and Alabama. In nearly all cases the state classifications have been linked or 'cross-walked' to NVC types, enabling some level of analysis at ecoregional or national scales. For more information on the use of state classifications and linkages to the NVC, contact your local natural heritage program.

Appendix C

Predictive Distribution Modeling

Development of a User-Interface PDM Modeling Tool

In June 2006 the Wyoming Natural Diversity Database and NatureServe completed initial development of a user-friendly software tool that guides modelers through the PDM process and automates many of the complex data processing, modeling, and mapping steps. According to Beauvais et al. (2006), this tool will be continually updated and maintained to stay current with new research results and modeling approaches. In this manner it will minimize the investment required to stay at the leading edge of technical literature, technological innovations, and conceptual advances. Ultimately the tool will implement a multiple modeling approach that will allows modelers to quickly apply several modeling techniques to particular occurrence data sets, then integrate the output from all approaches in summary maps with accurate and relevant evaluation measures. PDM is such a powerful approach to extracting information from raw occurrence data that it may become a standard practice within state natural heritage programs; this tool is an important first step in that direction.

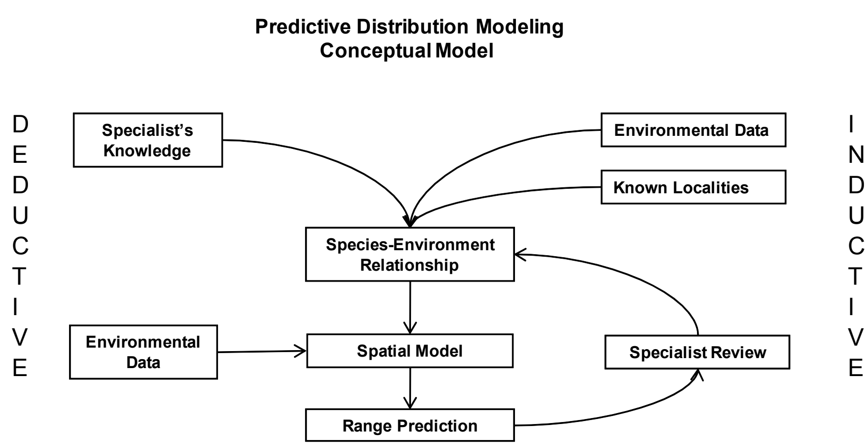
The most practical guidance regarding use of Predictive Distribution Modeling (PDM) for natural heritage type applications is provided in Beauvais et al. (2006). More theoretical aspects of PDM, as well as discussions of individual algorithms, are available in a number of recent publications, in particular, Guisan and Zimmermann (2000), Corsi et al. (2001), Ferrier et al. (2002), Scott et al. (2002), Elith and Burgman (2003), Rushton et al. (2004), and Elith et al. (2006). The summary provided below draws heavily from Beauvais et al. (2006) and is a broad overview of the techniques, strengths, and limitations of PDM.

Components of Predictive Distribution Models

There are three fundamental components of all PDM efforts: modeling (inductive or deductive), mapping, and evaluation. Each of these is discussed below.

Inductive and Deductive Models

Through 'inductive' modeling, PDM uses environmental predictors in correlation with known occurrences to spatially depict where that element might occur on the landscape. It is contrasted with 'deductive' modeling, a more subjective technique that uses known habitat affinities without relying on quantitative assessments of known locations (Figure E1). Quantitative data, which is most often used in 'inductive' modeling, will produce



a more detailed and accurate distribution map than qualitative data. However, models based on qualitative data and 'deductive' methods are still valuable when inductive modeling is not available. Furthermore, deductive methods may be used to augment inductive models when the number of occurrences used is particularly low.

Inductive modeling requires high quality environmental data of the study area and one or more known occurrences of the targeted element. It determines the strongest to the weakest relationships between predicting variables and the response variable. Not surprisingly, more occurrences and more detailed and finer scaled environmental information available for input result in a higher quality of output.

Vaughn and Ormerod (2003) divided environmental data into direct predictors and indirect predictors. Direct predictors have a direct biological relationship with the species of interest and include parameters such as temperature and pH. Elevation, topography, geology, and landform are examples of indirect predictors, which represent correlations with a series of intermediate direct factors. Strengths of direct variables include providing the user with a better understanding of the element-environmental relationship and improved information for conservation management. Strengths of indirect variables include increased overall efficiency of a model and broad availability of data for most areas.

Inductive models are most powerful and defensible when explanatory variables have been carefully selected by knowledgeable biologists to fit the element of interest. (Rushton et al. 2004).

There are a wide variety of inductive techniques currently used in PDM, including those that use only presence data (DOMAIN, GARP, BIOCLIM, and MaxEnt) and those that use both presence and absence data (Logistic regression, ordination, CART, RandomForest). No algorithm is the best for all applications; each has strengths and weaknesses that may be appropriate for particular modeling functions and data sets. DOMAIN, for instance, performs better than other models with small numbers of known occurrences (Elith et al 2006).

While inductive modeling techniques vary, a number of steps are common to all. The following steps were adapted from a NatureServe PDM project across a large landscape in Latin America for a rare bird species (Young, personal communication):

1. Select species.
2. Identify sources of localities (element occurrences from natural heritage/CDC source, museum specimens, or other sources).
3. Request and compile locality data.
4. Remove duplicate records.
5. Geo-reference and map localities.
6. First cut QC: remove/correct obvious errors.
7. Second cut QC: specialists review localities.
8. Select final input predictor variables.
9. Acquire and rectify environmental data.
10. Run trial models using appropriate algorithms.
11. Third cut QC: remove/correct outliers in environmental space.
12. Run final models.
13. Remove overpredicted area.

In contrast to inductive modeling, deductive modeling is conducted using qualitative information collected through literature surveys, expert interviews, or other data sources described in Section 4. As noted above, these models may be subjective and difficult to repeat with the same outcomes. However, this type of modeling is advantageous when models need to be produced quickly, there is minimal existing data, and/or resources are limited. Deductive modeling is not recommended for projects involving elements that have been poorly studied.

Mapping

Finished PDM products are usually in digital format. However, when hard copies are produced and used, three features are critical: background features are needed to orient the user, occurrence data points should be distinct and clearly visible, and desired output is appropriately displayed (transparent color schemes). More importantly, maps should be used with the disclaimer that PDM occurrences are merely predictions rather than known occurrences. A PDM map will often have an estimation of accuracy that is related to the scale of the input data.

Evaluation

Expert review, cross validation, and field surveys are used in model and map evaluation. Time and careful deliberation in the input phases typically result in a higher quality outputs: excellent models have been produced using very little data and average models have been produced using a moderate amount. Unfortunately, the quantity of existing data is often a direct result of the elements' rarity, with the rarest species sometimes lacking adequate data. In these cases, greater emphasis may need to be placed on the importance of negative data (the absence of a species). In all cases, the accuracy of the outputs should be reported by the modeler.

Quantitative evaluation. Without adequate data for cross-validation, (i.e., data points randomly held out from analysis and used for later validation), a model is considered an untested hypothesis. In other words, with what precision will the model place known occurrence in suitable environments and/or place known occurrence data outside of non-suitable environments? Model accuracy is often reviewed by using different configurations of the values in the "confusion matrix" (Guisan and Zimmermann 2000, Manel et al 2001), which assesses errors of both omission (predictions of absence where elements are likely to be) and commission (predictions of presence where elements are not likely to be).

Field evaluation. Ground-truthing is ultimately the best way to evaluate the accuracy of a predictive map. Because comprehensive field verification may be time consuming and expensive, it may be necessary to sub-sample a number of predicted locations for field surveys. In conducting field evaluations, it is important to search in both areas of predicted presence and absence.

Limitations of Predictive Distribution Modeling

The predicted distribution of an element is easily conveyed in a map, and therefore is a valuable deliverable of modeling projects. One of the most frequent misuses of distribution maps occurs when maps are mistakenly viewed as direct (or proven) instead of predicted relationships. Such use may be appropriate for immediate management actions that are risk-averse (e.g., citing a facility in an area predicted for species absence), but for most intentions field surveys should be employed to ascertain presence or absence on the ground before management decisions are made. Other limitations of PDM are described in Beauvais et al (2006) and briefly noted below:

Lack of GIS Data

Obviously, the ability to produce a useful map through PDM is reliant on high quality spatial data for the features used as predictors. Some potentially important information such as disturbance regimes, micro habitats, associated plant or animal species (e.g., host fish species for freshwater mussels) may not be available as spatial layers.

Sampling Bias

Inductive PDM may is often biased by uneven sampling efforts, particularly in models that use absence data (Vaughn and Ormerod 2003). Do blank areas on the map indicate species absence or just lack of inventories? A common survey bias is that in remote areas, greater sampling has been conducted in areas easily accessed by roads, which are likely to be at lower elevations and closer to water. Separating natural clustering from uneven sampling can be difficult but is often addressed by setting a minimum fixed distance between elements, with larger separation distances used with taxa that are more mobile (e.g. birds and some mammals).

Aquatic and Riparian elements

There are several unique challenges to modeling the distribution of aquatic elements. The distribution of individuals in a stream network may not reflect just the environmental features at those points of occurrence, but also environmental features and qualities of the drainage watershed (e.g., substrate pH, clay content). Moreover, it is hard to find consistent, high-resolution maps of water quality, streambed substrate, riparian features and other important aquatic features that drive the distribution of aquatic elements. Riparian environments tend to occur as thin strings or small patches that are often below the resolution of mapping projects, so many land cover maps do not show the true extent of riparian environments. In addition, many occurrence points have some level of error in mapping precision and thus have a greater tendency to map outside of thin riparian corridors. It is also a problem in the validation phase: validation points that come from observations within riparian corridors, but because of low mapping precision map outside of such corridors, will be scored as "misses" by the model when they were actually "hits". In Wyoming, some of these limitations have been addressed by a multiple modeling approach in which a statistical model (e.g., DOMAIN or CART), without any riparian or stream network information, is intersected with a buffered hydrology layer. The final map shows the buffered stream segments that occur within a generally suitable physical environment for that taxon. Another technique involves using distance-to-stream as a predictor variable in statistical models. As long as the points consistently fall close enough to streams to define a detectable statistical association, this appears to produce good models and resulting maps (Beauvais et al 2006).

Biogeographic Considerations

Unless they have rather small ranges, most elements probably do not use habitat consistently across their entire range (Dennis et al. 2003). Models of northern flying squirrels (Glaucomys sabrinus) from British Columbia, for example, may not apply to North Carolina because the climate, vegetation, soils, flora, fauna, and history of the two areas are very different. To model and map distribution across all of North America, it may be reasonable to include all known points of occurrence, from the Pacific to the Atlantic, but to model and map distribution only within British Columbia it would be inappropriate to include points from North Carolina, as the different habitat uses of North Carolina squirrels would mask the relevant patterns of British Columbian squirrels, and result in a poor predictive map. Consequently, it is logical to restrict distribution models to occurrence data from specific ecoregions.

In addition, there are practical constraints to extending a model outside of a particular study area. One common problem is that the spatial layers of predictor variables do not extend in a consistent fashion into adjacent areas, making it difficult or impossible to cross-walk different data sets into a single consistent layer. In these cases occurrence points from two or more states cannot be consistently attributed, which precludes development of a single, complete model from regional occurrence data.

In some cases, identifying suitable habitat outside of the native range of a taxon can be valuable to managers. For example, fisheries managers may be interested in knowing where particular species and subspecies of trout are most likely to thrive in a particular river or lake, even though these species are not native to that waterbody. While such uses of PDM are legitimate, the resulting products must be interpreted and used with the appropriate caveats.

Analyses Using Linear and Polygonal Features

Conventional PDM approaches assume that occurrence points are used as input, and each point is attributed with environmental values at the point center. There are equally valid biological reasons to use of polygonal and linear occurrences as input data (e.g., for large patch natural communities or riparian habitats). Increasingly, field observations are mapped as linear or polygonal features, and biologically-relevant syntheses of individual observations (like Element Occurrences) are often linear or polygonal features. The appropriate use of these features in PDM requires a thoughtful assessment of how to use the underlying environmental variables: average the attributes, use the dominant (spatially) attribute, or just use the polygon center?

Other Important Considerations for PDM

One of the most important factors of an effective model is the clear statement of model goals, functions, outputs, and audiences. Likewise, it is strongly desirable to use a model that incorporates transparent, realistic inputs and decisions rather than a 'black box' approach. Output users are more likely to use the model if they understand how it was made. Other key issues and considerations include:

Temporal Issues

Most PDMs are static and do not consider that the relationship between an element and its environment may change over time (Guisan and Zimmerman 2000). As a result, species that move in response to dynamic landscape processes (e.g., Canadian lynx, snowshoe hare, and forest condition in the northeast) may present challenges in modeling. Currently, changes in habitat over time (i.e., forest succession) need to be modeled by multiple PDM iterations. The intent to represent historic, current, or potential future distribution of an element influences the type of input data and analytical decisions.

Binary vs. Scaled Outputs

Are predictions of suitable vs. unsuitable occupation satisfactory (i.e., a binary model) or are degrees of suitable occupation desired for an occurrence (i.e., scaled model)? Models showing a gradient of likelihood are more likely to be realistic than models that portray only presence or absence.

Scale of Coverage

Model outputs will be limited by the scale of the inputs, and this scale should be appropriate for the model goals. For example, a multi-state model might have a coarser scale (1 km2 cell size) than a model for a particular landowner or national forest.

Evaluation and Validation

Practical evaluation (e.g., using known 'holdout' points that can provide independent validation, or better yet, field surveys of predicted positive and negative locations). Above all, model usefulness is dependant in the appropriate involvement and review of knowledgeable biologists.

Data Preparation

It is important not to underestimate the time required to prepare data for use, particularly if data are coming from multiple sources. The requirement of high quality data places added emphasis on the need for heritage programs to populate or update key data fields. Forest managers possess digital information at a high resolution (e.g., stand type, structure) that may be particularly useful in PDM.

Modeling Species with Few Occurrences

Specific guidance may be needed for G1 & G2 species or other elements which by nature have very few documented occurrences. Such guidance includes use of algorithms known to perform well with small population sizes (e.g., DOMAIN), use of species with very similar habitat preferences, and greater reliance on deductive techniques.

Physical vs. Biological Factors

PDM is most successful with physical variables that directly influence populations, since physical variables are typically more available in digital format than biological influences, such as predator/prey relationships and competitor species. Temporal issues, such as changing forest structure over time, would need to be assessed through multiple iterations in the model.

Appendix D

Ecological Community Sampling

Levels of Ecological Community Surveys

Variability in funding, staff capacity, terrain, and land access dictate that surveyors need to have flexibility to accommodate multiple levels of effort, or survey intensities, for ecological community surveys. Three levels are possible:

Level 1 = Remotely Sensed Measurement

Level 2 = Rapid or Extensive Evaluation (often qualitative)

Level 3 = Intensive Measurement and Evaluation (quantitative)

With increasing access to remotely sensed data, many metrics such as patch size, certain landscape context metrics, and structural attributes of abiotic/biotic condition may be readily addressed remotely. Consequently, Level 1 relies primarily on remotely sensed information. Level 2 typically involves a combination of remotely sensed data and rapid field assessment, including some quantitative field measures (e.g., 'reconnaissance' level inventory). This is the most common inventory form among natural heritage programs, and the expert judgment of natural heritage staff may play a strong role in the assessment of Level 2 data. Level 3 typically requires more intensive field-based assessment and may involve considerable quantifiable measurement (e.g., plot-based approach). Consideration of field sampling design for statistical purposes becomes most relevant in Level 3 assessment.

In many surveys some combination of Levels will be used. For instance, the area of a particular ecological association may be mapped using air photos (Level 1), a plant species list developed through a two-hour survey (Level 2), and quantitative tallies for forest structure derived by multiple plot samples (Level 3).

Ecological Community Sampling Design

Random Sampling

In standard random sampling each point has an equal probability of being sampled. Plot locations are randomly generated, and each plot may act as a sample unit. Sample units may then collectively or individually be analyzed to determine community composition and structure. Because this type of design does not attempt to reduce the effect of variability on desired estimates, a large sample size is typically needed to reach reasonable confidence intervals. In addition, randomly located plots are often inefficient in terms of field implementation. Thus, a completely random sampling design is seldom used because it is often not cost-effective. However, some amount of randomization is required to reduce bias and increase the accuracy of the estimated parameters, even if a design is not entirely random. Most statistical tests assume that the collection of observations is unbiased and independent (that is, selection of one observation has no influence on the selection of others). Although this assumption often cannot be achieved in the natural word, it is important to make an effort to avoid bias and collect samples with some randomization involved (Krebs 1989).

Systematic Sampling

Systematic sampling (i.e., along a transect or a grid) is often used to in increase the logistical efficiency, simplicity, and cost-effectiveness of sampling. In many cases a random starting point is selected sampling is repeated at a set distance along a transect thereafter. Because most ecological patterns are highly clumped and irregular, systematic sampling may result in an over-representation of common natural community types and an under-representation of uncommon or rare types.

Representative Sampling — Stratified Random Approach

In contrast to systematic sampling, representative sampling involves placing plots or points in proportion to the presumed coverage of a natural community, ecological system, or other ecological unit within a study area. Where insufficient information is known on ecological units, representative methods often rely on some inferred association between landscape structure and vegetation. A main weakness of this representative approach is that it relies too heavily on indirect factors (soils, landform, etc.) without explicitly stating the ecological relationships between vegetation and environment. Attempts to describe patterns in abiotic factors for efficient vegetation sampling design are most successful if the ecological meaning of the factors is understood (e.g. higher elevations are reflected in a transition from closed canopy hardwoods to stunted, open canopy conifers).

As noted above, random plot placement will not accurately reflect the full range of variability of the biotic and abiotic components of ecosystems at regional scales unless the sampling intensity is very high (Gauch 1982, Pielou 1984). To alleviate the shortcomings of standard random sampling, stratified sampling schemes have the potential to provide both accuracy in the recovery of patterns and statistical validity. Stratified sampling divides a study area into compartments and locates samples randomly within compartments. This approach has been used successfully over large heterogeneous areas with mostly unknown patterns. For example, a nested stratified random sampling design by landform and ecoregions was used in southern Yukon, Canada, to characterize vegetation pattern and its underlying environmental gradients (Orloci and Stanek 1979). The results of the study indicate that the selected stratifying variables accounted for a large part of the regional variation in vegetation.

Representative Sampling — Gradient-Oriented Transect (Gradsect) Approach

A primary goal of vegetation mapping and classification surveys is to characterize as many vegetation patterns as possible within the study area. The coverage of vegetation pattern is not necessarily accomplished by the usual statistical sampling procedures. Sampling theory emphasizes randomization in order to provide a probability structure for statistical analysis, but Gillison and Brewer (1985) argue that randomization may be counterproductive to the intent of ecological surveys because the occurrence of natural pattern is non-random. Data sets need to represent the full range of variability in biological patterns in response to variability in the environment.

Gradsect sampling is a variant of stratified random sampling that is based on the distribution of patterns along environmental gradients (Gillison and Brewer (1985). The gradsect sampling design (Gillison and Brewer 1985, Austin and Heyligers 1989) is intended to provide a description of the full range of biotic variability (e.g., vegetation) in a region by sampling along the full range of environmental variability. Transects that contain the strongest environmental gradients in a region are selected in order to optimize the amount of information gained in proportion to the time and effort spent during the vegetation survey (Austin and Heyligers 1989).

Helman (1983) and Austin and Heyligers (1989, 1991) expanded the gradsect methodology to include levels of environmental stratification within each gradsect. The procedure thus becomes a two-stage sampling design: (1) gradsects are chosen; (2) adequate environmental stratification and replication are performed within gradsects.

Tests of efficiency have shown that gradsects are generally more efficient than traditional statistical techniques in recovering the greatest amount of ecological pattern per sampling effort (Gillison and Anderson 1981; Gillison and Brewer 1985; Austin and Adomeit 1991). The gradsect method allows placement of samples in logistically more accessible areas than statistical techniques such as systematic or random sampling, thus improving cost-effectiveness. Costs can also be reduced and effectiveness maximized when stratifying variables are carefully chosen using existing information (Austin and Adomeit 1991).

Scale Considerations

In natural community inventories, complexity and spatial scale often complicate the relationship between classification and mapping. Plant communities form complex patterns on the landscape, each community representing a different combination of environmental conditions. At the scale of a small area, a map of plant associations may be used to characterize vegetation within the area itself. Even at this fine scale, however, plant communities may form complex mosaics composed of various phases (e.g., several ecological associations may intergrade within one large peatland ecological system). Vegetation associations that are mappable independently at the 1:24,000 scale may not be effectively mapped at 1:100,000 and smaller. At these scales (such as a midscale of 1:250,000), vegetation should be mapped as ecological systems. In this case, it is even more important to relate vegetation patterns to the ecological factors that shape them in order to be able to interpret the map units.

Plot Size (Area) and Shape

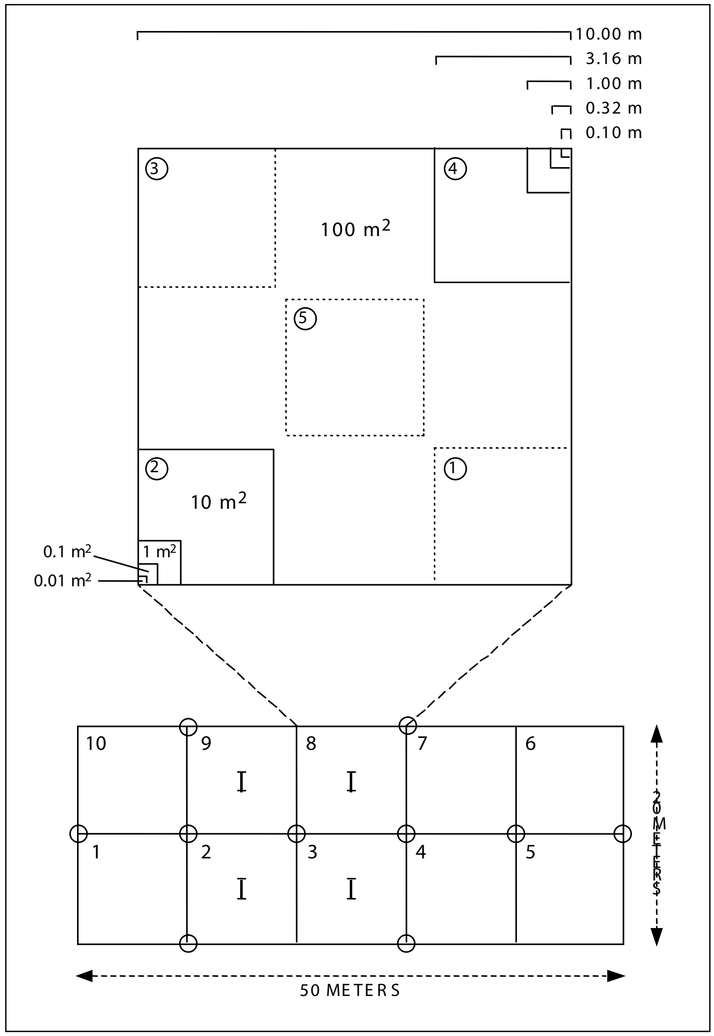
In determining the appropriate plot size needed to sample the vegetation, it is desirable to select a minimum area that will fully represent the species composition of the natural community. The minimal area can vary widely depending on the structure, scale of patterning, and species diversity of the community. The plot areas and dimensions below (Table D-1) may serve as useful guideline for selecting plot sizes (Mueller-Dombois and Ellenberg, 1974; Whittaker 1977). Sample plots should generally be rectangular in shape but may be varied to fit the nature of the occurrence (e.g., irregular plot shapes are used for seep communities that follow ravines).

TABLE D-1
Guidelines for Determining Plot Size (from National Park Service)

Class	Area	Dimensions
Forest	100 - 1,000 m^2	10x10 - 20x50
Woodland	100 - 1,000 m^2	10x10 - 20x50
Sparse Woodland	25 - 1,000 m^2	5x5 - 20x50
Shrubland	25 - 400 m^2	5x5 - 20x20
Sparse Shrubland	25 - 400 m^2	5x5 - 20x20
Dwarf Shrubland	25 - 400 m^2	5x5 - 20x20
Sparse Dwarf Shrubland	25 - 400 m^2	5x5 - 20x20
Herbaceous	25 - 400 m^2	5x5 - 20x20
Non-vascular	1 - 25 m^2	1x1 - 5x5

In some cases, a plot that is large enough to characterize the tree layer in a forest, woodland, or sparse woodland plot will be larger than is necessary to characterize the shrub or herb layer. In this case, nested subplots may be used to characterize the understory when this occurs (Figure D-1). The guidelines in Table D-1 may also be used to determine the appropriate size of the subplots (i.e., the herbaceous class guidelines may be used to determine subplot sizes appropriate for sampling the herbaceous layer of a forest community).

FIGURE D-1
Nested Plot Sampling Design Used in North Carolina (from Peet et al. 1997)



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