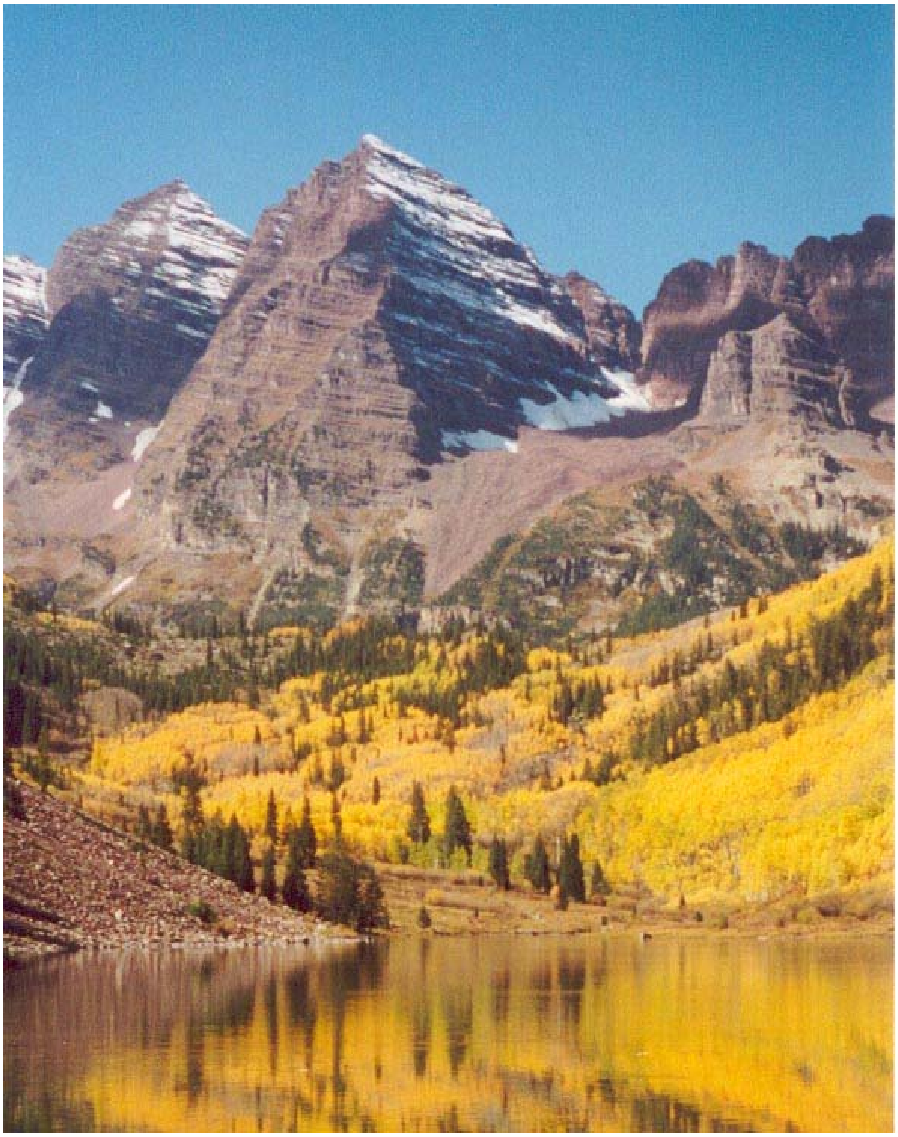
ECOLOGICAL SYSTEMS OF THE UNITED STATES

A WORKING CLASSIFICATION OF U.S. TERRESTRIAL SYSTEMS



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Ecological Systems of the United States

A Working Classification of U.S. Terrestrial Systems

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Executive Summary

Conservation of the Earth's diversity of life requires a sound understanding of the distribution and condition of the components of that diversity. Efforts to understand our natural world are directed at a variety of biological and ecological scales—from genes and species, to natural communities, local ecosystems, and landscapes. While scientists have made considerable progress classifying fine-grained ecological communities on the one hand, and coarse-grained ecoregions on the other, land managers have identified a critical need for practical, mid-scale ecological units to inform conservation and resource management decisions. This report introduces and outlines the conceptual basis for such a mid-scale classification unit—*ecological systems*.

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. They are intended to provide a classification unit that is readily mappable, often from remote imagery, and readily identifiable by conservation and resource managers in the field.

NatureServe and its natural heritage program members, with funding from The Nature Conservancy, have completed a working classification of terrestrial ecological systems in the coterminous United States, southern Alaska, and adjacent portions of Mexico and Canada. This report summarizes the nearly 600 ecological systems that currently are classified and described. We document applications of these ecological systems for conservation assessment, ecological inventory, mapping, land management, ecological monitoring, and species habitat modeling.

Terrestrial ecological systems are specifically defined as a group of plant community types (associations) that tend to co-occur within landscapes with similar ecological processes, substrates, and/or environmental gradients. A given system will typically manifest itself in a landscape at intermediate geographic scales of tens to thousands of hectares and will persist for 50 or more years. This temporal scale allows typical successional dynamics to be integrated into the concept of each unit. With these temporal and spatial scales bounding the concept of ecological systems, we then integrate multiple ecological factors—or *diagnostic classifiers*—to define each classification unit. The multiple ecological factors are evaluated and combined in different ways to explain the spatial co-occurrence of plant associations.

Summarizing across the range of natural variation, some 381 ecological systems (63%) are upland types, 183 (31%) are wetland types, and 35 (6%) are complexes of uplands and wetlands. Considering prevailing vegetation structure, 322 systems (54%) are predominantly forest, woodland, or shrubland, 166 systems (28%) are predominantly herbaceous, savanna, or shrub steppe, and 74 systems (12%) are sparsely vegetated or "barren."

Terrestrial ecological system units represent practical, systematically defined groupings of plant associations that provide the basis for mapping terrestrial communities and ecosystems at multiple scales of spatial and thematic resolution. The systems approach complements the U.S. National Vegetation Classification, whose finer-scale units provide a basis for interpreting larger-scale ecological system patterns and concepts. The working classification presented in this report will serve as the basis for NatureServe to facilitate the ongoing development and refinement of the U.S. component of an International Terrestrial Ecological Systems Classification.

Introduction and Background

Attempts to understand and conserve our natural world have often been directed at different biological and ecological levels, from genes and species, to communities, local ecosystems, and landscapes. Ecological conservation and resource managers typically require the identification, description, and assessment of some or all levels of biodiversity within a given planning area or ecoregion. Practically speaking, the *focal elements* that define these levels need to be clearly specified to clarify exactly what is to be protected or managed (Groves et al. 2002).

Conservationists and resource managers now use a variety of approaches to assess biodiversity at different scales (Redford et al. 2003). Species and ecoregions have received a great deal of attention. Species approaches include a focus on rare or endemic species, focal or umbrella species, and biodiversity hot spots. Ecoregional approaches include global prioritizations, such as the WWF Global 2000 ecoregions (Redford et al. 2003) or ecological land classifications (e.g., Albert 1995, Bailey 1996). Community and local ecosystem approaches have been less-well developed, though community approaches have been commonly used by natural heritage programs at the state level (e.g. Schafale and Weakley 1990, Reschke 1990). With the development of national and international vegetation classifications (Grossman et al. 1998, Rodwell et al. 2002, Jennings et al. 2003), the community approach is now applicable at more extensive geographic scales, at multiple levels of resolution. The local ecosystem approach has included mapping and assessment of fine-scaled landscape ecosystem units (e.g. see Barnes et al. 1998) or the definition of ecological system units within ecoregions (e.g. Neely et al 2001, Tuhy et al. 2002).

A common set of concerns for conservation or resource managers are: a) the spatial scale of the focal element (the "grain"); b) the degree of consistency in the element definition or taxonomy; c) the extent to which they can be applied across multiple jurisdictions or even continents; and d) the extent to which information can be readily assembled to assess their distribution, status, and trends. The species approach may require that grain be assessed on a species-by-species basis. The degree of consistency is improving as taxonomies improve, but parts of the world are not well surveyed. Worldwide lists and red lists are increasingly available, but information on many species is often difficult to obtain.

Ecoregional approaches often provide multiple levels of spatial scales, but typically the grain is quite coarse, and the units are unique subsets of the geographic space, with varying degrees of heterogeneity. They are either used as focal elements directly or as organizing units for focusing on more specific focal elements within the region. They are now increasingly available around the world, and information can be readily assembled, depending on the features of the ecoregion being assessed.

Community approaches, often considered a more convenient focal element (the "coarse filter'), as compared to species (the "fine filter") (Jenkins 1976), often have a fine grain, are relatively consistent, but are often not feasibly applied to national or broader assessments (e.g. Noss and Peters 1995). Their fine grain may hinder ability to assemble information and conduct assessment, limiting their practical value. Our experience in the application of the International Vegetation Classification (IVC) and its U.S. component, the U.S. National Vegetation Classification[1] (NVC) has indicated the need for standardized classification units that more fully integrate environmental factors into unit definition (e.g. Anderson et al. 1999). There is also a need to define units somewhat more broadly than individual NVC floristic units (alliances and associations) – i.e., allowing for a greater range of biotic and abiotic heterogeneity in type definition – without "scaling up" to the NVC formation unit, which is defined solely through vegetation physiognomy and limited environmental factors.

Finally, the intermediate-scaled landscape ecosystems (e.g. USFS ECOMAP Land Type Associations) are often difficult to define consistently, and may be rather heterogeneous with respect to biodiversity. They are not fully developed or widely available across the country, or across continents, making it difficult to use these units in regional, national, or international assessments.

Lacking in these approaches is a focal element that is more coarsely grained than the community approach, retains a standard of consistency that allows ready identification and application of the unit at local or regional scales, and that is widely applicable at continental or hemispheric levels. In addition, gathering information on such focal elements should not make excessive information demands on conservation or resource managers. Here we describe a standardized terrestrial ecological system classification designed to meet these objectives. Our purpose is to demonstrate that these systems, though related to both community and landscape ecosystem approaches, provide a greatly improved set of focal elements for conservation and resource management.

Ecological Scope of Classification. The emphasis of this classification is directed towards surficial terrestrial environments, encompassing both *upland* and *wetland* areas where rooted and non-vascular vegetation – as well as readily identifiable environmental features (e.g. alpine, coastal, cliff, sand dune, river floodplain, depressional wetland, etc.) - may be used to recognize and describe each type. We do not address either subterranean environments, or aquatic environments, whether freshwater or marine. Within terrestrial environments, we focus here on existing ecological system types that can be considered "natural" or "near-natural," i.e., those that appear to be unmodified or only marginally impacted by human activities. This is to provide a framework for describing ecological composition, structure, and function that has existed with minimal human influence under climatic regimes of recent millennia. We have made no attempt to classify and describe agricultural ecosystems or urban ecosystems where human-caused elements

[1] See Appendix 1 for further explanation of the U.S. National Vegetation Classification as well as other existing classification approaches.

are clearly novel in a temporal context of 100s to 1000s of years. Instead, as we apply this classification to mapping, we rely on broadly based land cover classes to identify and map human-dominated areas. With this approach, we are still able to track the current status of natural ecosystems relative to cultural ones, and even suggest how human alterations may be viewed more directly in light of presumed historical conditions.

Geographic Scope of Classification. NatureServe is currently working toward a first-draft classification of terrestrial ecological systems across North and South America –an International Terrestrial Ecological Systems Classification. A team of NatureServe and natural heritage program ecologists has now completed a working list and descriptions of the U.S. Terrestrial Ecological Systems Classification, which includes nearly 600 terrestrial ecological systems in the coterminous, lower 48 United States, portions of southern coastal Alaska, and ecologically similar regional landscapes in adjacent southern Canada and northern Mexico (Figure 1). Their distribution by ecoregions, as defined by The Nature Conservancy (Groves et al. 2002), is also documented, thereby providing a list of focal elements that can facilitate conservation work in that organization.

The Iterative Nature of Classification. Ecological classifications, such as this one, should be viewed as an ongoing process of stating assumptions, data gathering, data analysis and synthesis, testing new knowledge through field application, and classification refinement. A classification system provides a framework for this ongoing process and the resulting classification should continually change as new knowledge is gained. The effort documented here represents the first attempt to synthesize data and apply a standard approach to documenting natural upland and wetland ecological systems comprehensively across the coterminous United States. Although in this report we include adjacent regions based on the ecoregional boundaries that extend beyond the U.S., additional collaboration with partners is needed to advance this classification internationally. NatureServe will continue to provide a mechanism for ongoing development and dissemination of this classification.

Objectives of This Report. This report documents the development of terrestrial ecological systems, emphasizing the key issues and requirements of such a system in relation to other approaches. We review the criteria used to classify systems and the standards that were used to develop, name, and describe them. We describe the process for gathering information on these systems and summarize the results of this initial classification effort. We then describe the application of ecological system units for mapping and assessing occurrence quality or ecological integrity. We also describe the application of these units to conservation assessment and description of wildlife habitat. Finally we address the next steps in the process of further enhancing the systems classification.

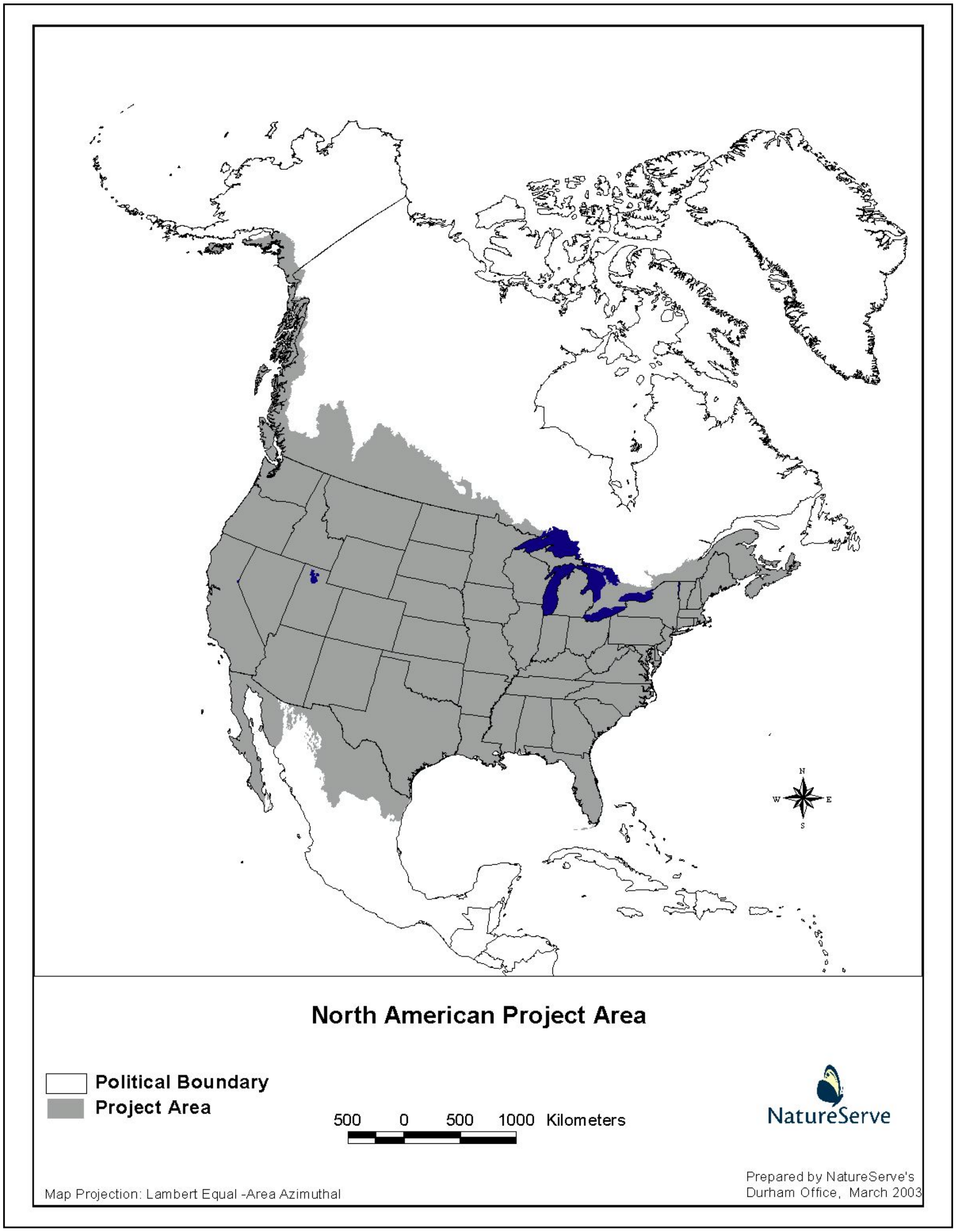


Figure 1. Project Area included in this classification effort.

Key Issues and Decisions in Developing Ecological Systems

Ecosystems have been defined generally as " a community of organisms and their physical environment interacting as an ecological unit" (Lincoln et al. 1982). Classification of ecological systems can be based on a variety of factors (e.g., vegetation, soils, landforms) at a variety of spatial and temporal scales (hectares to millions of kilometers and annual to millennial), and with varying degrees of concern over spatial interactions. A full review of the variety of classifications currently used is beyond the scope of this document. Rather, some key issues will be highlighted that includes discussions of other approaches. See Appendix 1 for a review of some major classifications that informed our approach.

Ecological Systems as Functional Units versus Landscape Units

Historically, ecological systems have been defined from a wide variety of perspectives, depending on the investigator. Some have emphasized the "physical" (land) factors that structure the system; others have emphasized ecosystem function and processes, such as nutrient cycling and energy flows (Golley 1993). Odum (2001) emphasizes the latter perspective in his definition of ecological system:

> *An ecological system, or ecosystem, is any unit (a biosystem) that includes all the organisms (the biotic community) in a given area interacting with the physical environment so that a flow of energy leads to clearly defined biotic structures and cycles of materials between living and non-living parts. An ecosystem is more than a geographic unit (or ecoregion); it is a functional system with inputs and outputs, and with boundaries that can be neither natural or arbitrary.*

The emphasis is on energy flow and nutrient cycling, looking at how primary and secondary producers shape the flow of energy and materials through a system. By contrast, Bailey (1996) emphasizes the landscape ecosystem approach:

J. S. Rowe … defined an ecosystem as "a topographic unit, a volume of land and air plus organic contents extending areally over a particular part of the earth's surface for a certain time." This definition stresses the reality of ecosystems as geographic units of the landscape that include all natural phenomena and that can be identified and surrounded by boundaries."

These definitions do not lead to mutually exclusive approaches to ecosystem studies. Many functional studies use watershed geographic units to define their ecosystems; and landscape ecosystem studies often emphasize functional properties within and across geographic units. Our decision was to emphasize a classification approach to ecosystems that does not rely on a fixed landscape map unit and which is still amenable to process-functional studies. We emphasize how processes on the landscape shape ecological systems, and define them through a combination of biotic and abiotic criteria.

Ecological Systems as Geo-Systems versus Bio-Systems

Given that ecosystems generally are defined as an ecological unit of both organisms and their environment, there are various approaches to choosing which set of factors to emphasize in a classification. The landscape ecosystem, or geo-ecosystems (Rowe and Barnes 1994), emphasizes the controlling factors of climate, soils, and topography over that of biota. The bio-ecosystems approach gives more emphasis to the controlling factors of biota (akin to the “biogeocoenosis” of Sukachev 1945, in Mueller-Dombois and Ellenberg 1974, or the biogeocene unit of Walter 1985).

The bio-ecosystem approach has recently received more widespread attention for conservation and resource management through the development of “biotope” units. A **biotope** (sometimes called “habitats”) is a small to meso-scale ecosystem unit, defined as “a limited geographic area with a particular environment and set of flora and fauna” (Devillers et al. 1991). In Europe, habitat types have been defined at a variety of scales by the CORINE Biotope Manual, which defined and described hundreds of habitat types (Devillers et al. 1991). But, due to ambiguity in the definition of these units, a more recent EUNIS habitat list was published (Davies and Moss 1999), which was explicitly tied to plant communities (alliances) of the Braun-Blanquet school (Rodwell et al. 2002). In this way the boundaries of the system could be more clearly recognized through their component plant communities.

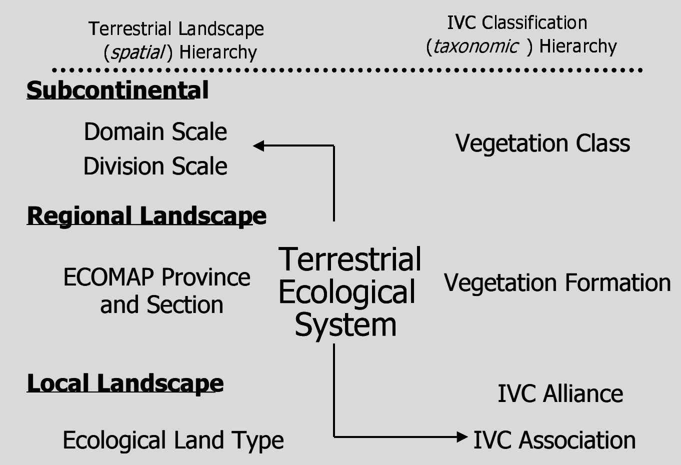
Our decision was to define ecological systems using a “bio-ecosystem” approach. We also chose to classify these systems at a meso-scale (akin to the “biogeocene complex” unit of Walter 1985). This approach defines the boundaries of a system in part based on the combination of component plant communities and abiotic factors. We chose to link our system units to the plant communities defined in the IVC / USNVC (Grossman et al. 1998) as a way of explicitly defining the boundaries of the system. The vegetation units are based on existing vegetation, and so our systems are also based on “existing ecosystems,” not potential systems.

Nonetheless, the geo-ecosystem approach has an important role to play in helping define the abiotic template on which ecological systems may be found. Geo-ecosystem ecological land units (ELUs), such as the ecological land types of the ECOMAP hierarchy, or the ecosite types of various Canadian FECs[2], can play an important role in the predictive modeling of ecological systems, where the abiotic factors that define our systems can be linked to those used to define ELUs.

2 See e.g., Racey et al. (1996) for northwestern Ontario. Canadian FEC ecosites vary from province to province, and in some cases, these ecosites may be more-or-less equivalent to our ecological system concept.

Our approach may reinforce the notion that ecological systems are always broader than individual communities (e.g., an ecological hierarchy that proceeds from populations to communities to ecosystems to landscapes) (King 1993). We recognize that, in general, communities and ecosystems are not defined *a priori* in terms of these relationships – communities could be defined at broader scales than ecosystems (such as the temperate broadleaf forest *Formation* of the IVC as compared to one of our forest system units, or a rotting log ecosystem within a beech-maple forest association). Rather, our approach to defining ecological systems at particular spatial and temporal scales would typically encompass a number of community types defined at the scales of the IVC/NVC floristic units *(association or alliance),* or for that matter the finer-scaled landscape ecosystem units defined by ECOMAP (see Box 2). Our reasons for doing so are pragmatic. We see a need for such a meso-scale unit that is not available in either of those hierarchies.

Box 1:
Linking Ecological Systems to other Hierarchies



Ecological Systems as Discrete Units versus Individualistic Units

Whether as bio-ecosystems or geo-ecosystems, the concept of ecological systems can be rather ambiguous (King 1993). Because geo-ecosystems are often portrayed as maps, they may appear as fairly discrete units, but this is more a reflection of the mapping process than the inherent discreteness of the units. Debate over the relative discreteness of ecosystem types parallels a similar debate in vegetation ecology. The "continuum concept" in vegetation, as developed by Gleason (1926), Curtis (1959), and Whittaker (1956, 1962) argues that because species have individual, independent responses to the environment, their individualistic response produces a continuum of change along gradients. This concept reflects, as well, the individualistic nature of the environment: no two segments of the physical terrain are identical. The issue for vegetation applies equally to ecosystems. The debate between those holding the continuum view and those supporting the "community unit concept" (see Clements 1916, Daubenmire 1966)—which held that communities recur consistently and are successionally directed toward stable "climax" conditions—has led to a consensus that, in general, the continuum concept offers a realistic view of natural pattern in terrestrial environments (McIntosh 1993). However, there is also ample recognition that species and habitats found in a given area are structured to some degree by interactions with each other, their environment, disturbance regimes, and historical factors, and many combinations of species and habitats do indeed recur (e.g., Austin and Smith 1989). This viewpoint – one that is perhaps intermediate between the "community unit concept" and the "continuum concept" – has been widely used in guiding ecological classification. Although there is continuous variation in species composition and environmental gradients, in some places the level of compositional and environmental change is low (e.g., within a readily recognizable plant community) whereas in other places the level of compositional change is high (e.g., across an ecotone).

The necessary expression of these findings is that in most cases there are no unambiguous boundaries between plant communities or ecological systems in nature, and species assemblages or ecosystem processes are not entirely predictable. Any method of dividing the continuously varying and somewhat unpredictable phenomenon of community types and systems must be somewhat arbitrary with multiple acceptable solutions. Ecological classification only requires that it is reasonable to separate the continuum of variation in ecological composition and structure into a series of somewhat arbitrary classes (Whittaker 1975, Kimmins 1997). Furthermore, ecosystem factors are typically more temporally and spatially stable than vegetation factors on their own, facilitating repeated recognition of the same unit.

We recognize that ecological systems do grade more-or-less continually across the landscape. We rely on a combination of diagnostic classifiers of both abiotic and biotic factors to create reasonable classes of units. We further incorporate plant community types already defined in the NVC to help place boundaries on the system units.

The Scale of Ecological Systems

In principle, ecosystems can be defined at any geographic scale, from a rotting log or vernal pond to the entire biosphere. Typically they range from <10 to 1,000,000s of hectares. They can also vary in the definition of their stability, from annual to 1,000s of years (Delcourt and Delcourt 1988). Recent classifications or regionalizations using the geo-ecosystem approach explicitly define a nested series of spatial scales, from broad-ranging ecoregional units that span millions of hectares to "micro-ecosystem" land types that span 10s of hectares. The expectation is that these units are stable on the order of hundreds of years. Functional approaches work at a variety of temporal and spatial scales as well, depending on the processes being studied.

In developing this ecological systems classification, we decided to focus on the scale of greatest need. Good classifications exist at both the micro- and macro-ecosystem level; for micro-ecosystems, there are either the plant community associations of the NVC (Grossman et al. 1998, NatureServe 2003, Jennings et al. 2003) or the ecological land types of ECOMAP (Bailey 1996). Spatially, these micro-ecosystems are usually defined at scales of 10s to 1,000s of hectares. Temporally the associations typically reflect vegetation stability at scales of 10 to 100 or more years; the ecological land type also typically emphasize soil-landform stability at the scale of 50 to 100s of years. At macro-ecosystem scales, vegetation formations (UNESCO 1973, FGDC 1997, Grossman et al. 1998) or ecoregions (Bailey 1996) can be used. Spatially, these macro-systems often span continents. Temporally, formations vary in their stability (though recognition tends to focus on the more stable units), and ecoregions emphasize stability on the order of 100s to 1000s of years.

Notably lacking, however, are good meso-scale units. For bio-ecosystems that rely on plant communities, the change in scale between formations and alliance units is rather large. Experience in application of the NVC has indicated the need for units that are somewhat more broadly defined than individual NVC alliance and association units – i.e. allowing for a greater range of biotic and abiotic heterogeneity in type definition – without "scaling up" to the NVC formation unit, which is defined solely through vegetation physiognomy and limited environmental factors. For geo-ecosystems, the meso-scale units of subsections and land type association units are still in development, and standards are still lacking across the country (Smith 2002).

Thus, our decision was to focus on meso-scale ecological system units. The problem we are addressing is not new. Walter (1985, p. 17) stated:

> *Between the biomes on the one hand and the biogeocenes [corresponding to the plant community with the rank of an association], on the other, is a wide gap, which has to be filled by units of intermediary rank. These units we propose to call biogeocene complexes. They often correspond to a particular kind of landscape, have a common origin, or are connected with one another by dynamic processes. As an example, we can cite a biogeocene sequence on a slope with lateral material transport (catena) or a natural succession of biogeocenes in a river valley or a basin with no outlet…The different types have as yet been given no ecological names of their own…*

In conclusion, our approach to classifying ecological systems draws from a variety of previous efforts to define ecological units, whether as plant community types or ecological land types. We determined that a consistent meso-scale ecosystem that could span the North and South American continents was missing from available classification approaches. We focused our efforts on developing such a unit, one that could address basic patterns of ecological variability and serve to guide conservation and resource management needs.

Terrestrial Ecological Systems: Conceptual Basis

A terrestrial ecological system is defined as a group of plant community types that tend to co-occur within landscapes with similar ecological processes, substrates, and/or environmental gradients. A given terrestrial ecological system will typically manifest itself at intermediate geographic scales of 10s to 1,000s of hectares and persist for 50 or more years.

Ecological processes include natural disturbances such as fire and flooding. Substrates may include a variety of soil surface and bedrock features, such as shallow soils, alkaline parent materials, sandy/gravelling soils, or peatlands. Finally, environmental gradients include local climates, hydrologically defined patterns in coastal zones, arid grassland or desert areas, or montane, alpine or subalpine zones.

By plant community type, we mean a vegetation classification unit at the association or alliance level, where these are available in the International Vegetation Classification (IVC) and its U.S. component, the USNVC (NVC) (Grossman et al. 1998, Jennings et al. 2003, NatureServe 2003), or, if these are not available, other comparable vegetation units. NVC associations are used wherever possible to describe the component biotic communities of each terrestrial system. The NVC provides a multi-tiered, nested hierarchy for classifying vegetation types. Currently the NVC includes over 5,000 vegetation associations and 1,800 vegetation alliances described for the coterminous United States.

Ecological systems are defined using both spatial and temporal criteria that influence the grouping of associations. Associations that consistently co-occur on the landscape therefore define biotic components of each ecological system type. Our approach to ecological systems definition using IVC associations is similar to the biotope or habitat approach used, for example, by the EUNIS habitat classification, which explicitly links meso-scale habitat units to European Vegetation Survey alliance units (Rodwell et al. 2002). Given the relative ease of recognizing vegetation structure and composition, this approach is preferable to defining biotic components using animal species that are more difficult to consistently observe and identify.

In developing an ecological systems approach, we are mindful that ecological systems can be defined in a number of ways. Indeed, there are so many different definitions that some have suggested that the concept is in danger of losing its utility. O'Neill (2001) made a number of suggestions to help improve the ecosystem concept: that the ecosystem (1) be explicitly scaled, (2) include variability, (3) consider long-term sustainability in addition to local stability, and (4) include population processes as explicit system dynamics. We define our ecological system concept as follows:

1. We explicitly scale the unit to represent, in most cases:
    a. spatial scales of tens to thousands of hectares
    b. temporal scales of 50 to 100 years
2. We make explicit the variability in the system by describing them in terms of a consistent list of abiotic and biotic criteria and by linking ecological systems to plant community types (associations and alliances of the NVC) that describe the biotic community variation within the system.
3. We propose to consider long-term sustainability and local stability by mapping and evaluating the occurrence of ecological systems at the local site and the regional level.
4. We do not formally include population processes as explicit system dynamics, but through knowledge of the component plant communities, we are at least able to describe the major plant species and their dynamics within the systems. Additional work could formalize the roles of additional biotic elements such as invertebrates and vertebrates.

Meso-Scale Ecosystems

Our concept of terrestrial ecological systems includes temporal and geographic scales intermediate between stand and landscape-scale analyses. These “meso-scales” constrain the definition of system types to scales that are of prime interest for conservation and resource managers who are managing landscapes in the context of a region or state. More precise bounds on both temporal and geographic scales take into account specific attributes of the ecological patterns that characterize a given region.

Temporal Scale: Within the concept of each classification unit, we clearly acknowledge the dynamic nature of ecosystems over short and long-term time frames. If we assumed that characteristic environmental settings (e.g. landform, soil type) remain constant over the time period that applies to ecological systems (fifty to several hundred years), we would still encounter considerable within-system variation in vegetation due to disturbance and successional processes. Our temporal scale determines the means by which we account for both successional changes and disturbance regimes in each classification unit. Relatively rapid successional changes resulting from disturbances are encompassed within the concept of a given system unit. Therefore, daily tidal fluctuations will be encompassed within a system type. Some of the associations describing one system may represent multiple successional stages. For example, a given floodplain system may include both early successional associations and later mature woodland stages that form dynamic mosaics along many kilometers of a river. Many vegetation mosaics resulting from annual to decadal changes in coastal shorelines will be encompassed within a system type. Many forest and grassland systems will encompass common successional pathways that occur over 20-50 year periods. Selecting this temporal scale shares some aspects with the "habitat type" approach to describe potential vegetation (Daubenmire 1952, Pfister and Arno 1980), but differs in that no "climax" vegetation is implied, and all seral components are explicitly included in the system concept.

Of course, many environmental attributes, such as climate, continually change over much longer and more varied time frames. Our concept for any "natural/near-natural" ecological system type encompasses temporal variation that is responding to climatic variations that have occurred in recent millennia, with little or no human influence.

Pattern and Geographic Scale: Spatial patterns that we observe at "intermediate" scales can often be explained by landscape attributes that control the location and dynamics of moisture, nutrients, and disturbance events. For example, throughout temperate latitudes one can often see distinctions in vegetation occupying south-facing vs. north-facing slopes or from ridge top to valley bottom. Site factors in turn may interact with insect, disease, and fire. Another example can be taken from floodplains. Rivers provide moisture, nutrients, and soil disturbance (scouring or deposition) that regulate the regeneration of some plant species. In these settings we find a number of associations co-occurring due to controlling factors in the environment. We see mosaics of associations from different alliances and formations, such as woodlands, shrublands, and herbaceous meadows, occurring in a complex mosaic along a riparian corridor. Some individual associations may be found in wetland environments apart from riparian areas. But we can often predict that along riparian corridors within a given elevation zone, and along a given river size and gradient, we should encounter a limited suite of associations. It is these "meso" spatial scales that we address using ecological systems.

Diagnostic Classifiers

As the definition for ecological systems indicates, this is a multi-factor approach to ecological classification. Multiple environmental factors—or *diagnostic classifiers*—are evaluated and combined in different ways to explain the spatial co-occurrence of NVC associations (Box 2). Diagnostic classifiers is used here in the sense of Di Gregorio and Jansen (2000); that is, the structure of the ecological systems classification is more "modular" in that it aggregates diagnostic classifiers in multiple, varying

Box 2: Diagnostic Classifiers
(Categories and Examples)

Ecological Divisions
- Continental Bioclimate and Phytogeography

Bioclimatic Variables
- Regional Bioclimate

Environment
- Landscape Position, Hydrogeomorphology
- Soil Characteristics, Specialized Substrate

Ecological Dynamics
- Hydrologic Regime
- Fire Regime

Landscape Juxtaposition
- Upland-Wetland Mosaics

Vegetation
- Vertical Structure and Patch Type
- Composition of component associations
- Abundance of component association patches

combinations. Instead of a specific hierarchy, we present a single set of ecological system types. This is in contrast to, for example, the framework and approach of the IVC. The nested IVC hierarchy groups associations into alliances based on common dominant or diagnostic species in the upper-most canopy. This provides more of a taxonomic aggregation with no presumption that associations within the alliance co-occur in a given landscape. The ecological system unit links IVC associations using multiple factors that help to explain why they tend to be found together in a given landscape. Therefore, ecological systems tend to be better "grounded" as ecological units than most IVC alliances and are more readily identified, mapped, and understood as practical ecological units. Diagnostic classifiers include a wide variety of factors representing bioclimate, biogeographic history, physiography, landform, physical and chemical substrates, dynamic processes, landscape juxtaposition, and vegetation structure and composition.

Biogeographic and Bioclimatic Classifiers. Ecological Divisions are sub-continental landscapes reflecting both climate and biogeographic history, modified from Bailey (1995 and 1998) at the Division scale (Figure 2). Continent-scaled climatic variation, reflecting variable humidity and seasonality (e.g. Mediterranean vs. dry continental vs. humid oceanic) are reflected in these units, as are broad patterns in phytogeography (e.g. Takhtajan 1986). The division lines were modified by using ecoregions established by The Nature Conservancy (Groves et al. 2002) and World Wildlife Fund (Olson et al. 2001) throughout the Western Hemisphere. These modified divisional units aid the development of system units because regional patterns of climate, physiography, disturbance regimes, and biogeographic history are well described by each Division. Thus, these divisions provide a starting point for thinking about the scale and ecological characteristics of each ecological system. Examples of these Divisions include the Inter-Mountain Basins, the North American Warm Desert, the Western Great Plains, the Eastern Great Plains, the Laurentian and Acadian region, the Rocky Mountains, and the Atlantic and Gulf Coastal Plain. A "Rocky Mountain" ecological system type is entirely or predominantly found (>80% of its total range) within the Rocky Mountain Division. A "Southern Rocky Mountain" ecological system type is limited in distribution to southern portions of the broader Rocky Mountain Division. In a few instances, ecological systems remain very similar across two or more Ecological Divisions. In these instances, the Domain scale of Bailey (1998) was used to name and characterize the distribution of types; e.g. the "North American Arid West Emergent Marsh" spans the North American Dry Domain.

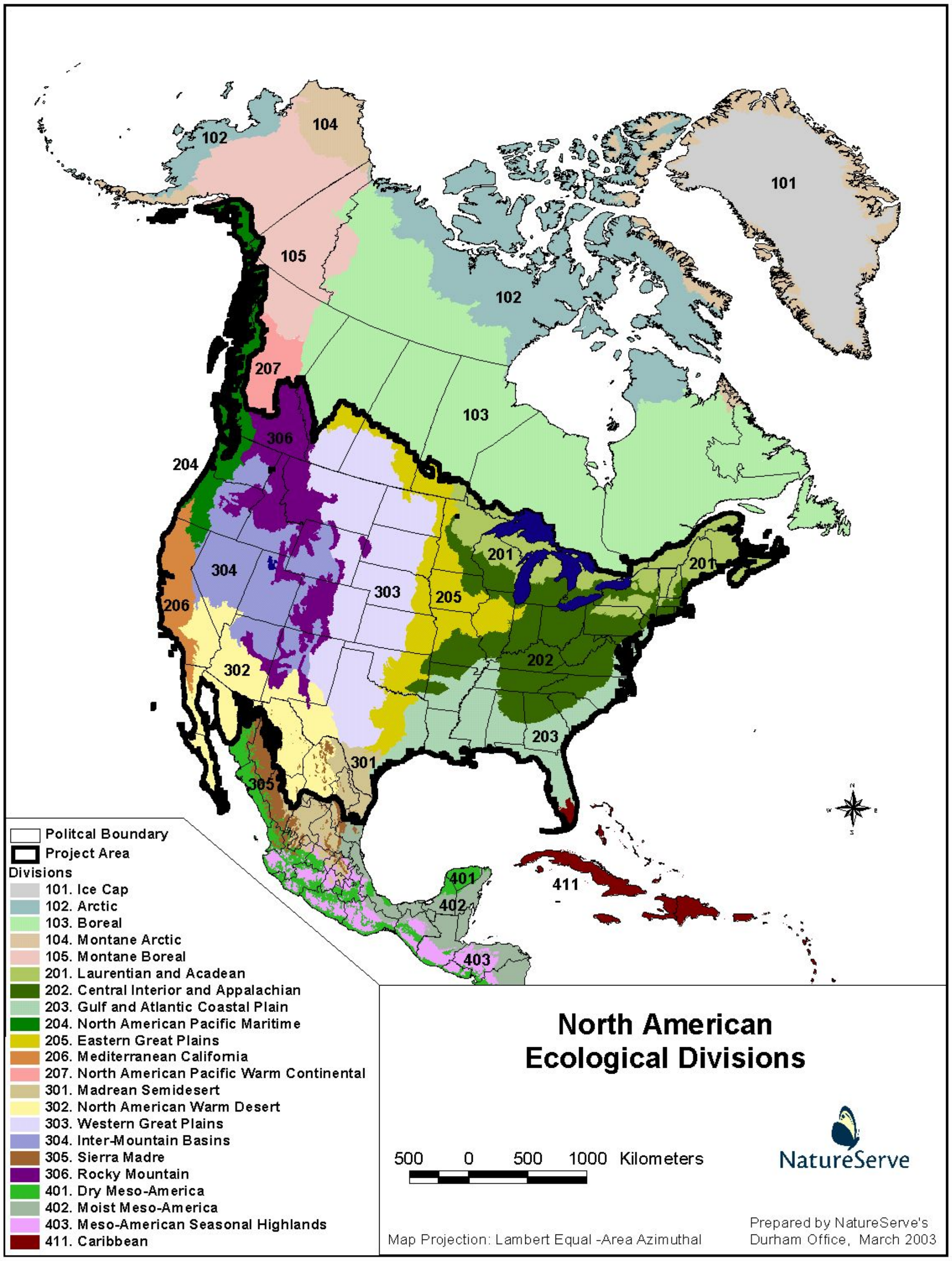


Figure 2. Ecological Divisions of North America used in organization and nomenclature of NatureServe Ecological Systems. Project area of this report is highlighted.

Subregional bioclimatic factors are also useful for classification purposes, especially where relatively abrupt elevation-based gradients exist, or where maritime climate has a strong influence on vegetation. We integrated global bioclimatic categories of Rivas-Martinez (1997) to characterize subregional climatic classifiers. These include relative temperature, moisture, and seasonality. They may be applied globally, so they aid in describing life zone concepts (e.g. 'maritime,' 'lowland,' 'montane,' 'subalpine,' 'alpine') in appropriate context from arctic through tropical latitudes.

Environment: Within the context of biogeographic and bioclimatic factors, ecological composition, structure and function in upland and wetland systems is strongly influenced by local physiography, landform, and surface substrate. Some environmental variables are described through existing, standard classifications and serve as excellent diagnostic classifiers for ecological systems. For example, soil moisture characteristics have been well described by the Natural Resource Conservation Service (NRCS 1998). Practical hydrogeomorphic classes are established for describing all wetland circumstances (Brinson 1993). Other factors such as landforms or specialized soil chemistry may be defined in standard ways to allow for their consistent application as diagnostic classifiers.

Ecological Dynamics. Many dynamic processes are sufficiently understood to serve as diagnostic classifiers in ecosystem classification. In many instances, a characteristic disturbance regime may provide the single driving factor that distinguishes system types. For example, composition and structure of many similar woodland and forest systems are distinguishable based on the frequency, intensity, periodicity, and patch characteristics of wildfire (Barnes et al. 1998). Many wetland systems are distinguishable based on the hydroperiod, as well as water flow rate, and direction (Brinson 1993; Cowardin 1979). When characterized in standard form (e.g. Frost 1998), these and other dynamic processes can be used in a multi-factor classification.

Landscape Juxtaposition. Local-scale climatic regime, physiography, substrate, and dynamic processes can often result in recurring mosaics. For example, large rivers often support recurring patterns of levee, floodplain, and back swamps, all resulting from seasonal hydrodynamics that continually scour and deposit sediment. Many depressional wetlands or lakeshores have predictable vegetation zonation driven by water level fluctuation. The recurrent juxtaposition of recognizable vegetation communities provides a useful and important criterion for multi-factor classification.

Vegetation Structure, Composition, and Abundance: As is well recognized in vegetation classification, both the physiognomy and composition of vegetation suggests much about ecosystem composition, structure, and function. However, the relative significance of vegetation physiognomy may vary among different ecosystems, especially at local scales. For example, many upland systems support vegetation of distinct physiognomy in response to fire frequency and soil moisture regimes. In general, physiognomic distinctions such as "forest and woodland," "shrubland" "savanna," "shrub steppe," "grassland, " and "sparsely vegetated" are useful distinctions in upland environments. On the other hand, needleleaf or broadleaf tree species that are either evergreen or deciduous may co-occur in various combinations due more to variable responses to natural disturbance regimes or human activities than to current environmental conditions. Many wetland systems could support herbaceous vegetation, shrubland, and forest structures in the same location, again, based on the particular strategies of the species involved and local site history.

Therefore, while recognizable differences in vegetation physiognomy may initially suggest distinctions among ecosystem types, knowledge of vegetation composition should be relied upon more heavily to indicate significant distinctions. As in vegetation classification, we recognize beta diversity, or the turnover of species composition through space, as a primary means of differentiating ecosystem types. The task of classification is to recognize where that turnover is relatively abrupt, and to explain why that abrupt change occurs on the ground.

Standarized vegetation classifications, especially at the local scale described by the NVC association concept, provide a useful tool for qualitative evaluation of vegetation similarity among ecological systems. In locations where NVC associations are well developed, they serve as a useful summary of quantitative data on the physiognomy and floristics of vegetation across the United States. For example, two apparently similar forest ecosystems could be characterized in terms of the NVC associations they support. We can assess the relative similarity of the two systems by comparing the association lists. Of course, detailed and comprehensive association-scale classification is not always available, especially in subtropical and tropical regions. In these instances, qualitative description and evaluation of non-standard classification units is often sufficient for initial characterization of vegetation physiognomy and composition among ecological systems.

While beta diversity is a primary consideration, the relative abundance of vegetation can also be an important consideration. For example, riparian and floodplain systems may share many plant species, due to their adaptation for dispersal along a seasonally flowing river. However, there may be substantial differences in the relative abundance of vegetation between, for example, riparian systems with small, flash-flood stream dynamics and a large, well-developed river floodplain many kilometers downstream. Measurement of both vegetation patterns and environmental factors that support them are needed to adequately address this facet of ecological classification.

Methods of Classification Development

Ideally, ecological classification proceeds through several phases in a continual process of refinement. These phases could include: 1) literature review and synthesis of current knowledge; 2) formulating an initial hypothesis describing each type, that supports; 3) establishing a field sampling design; 4) gathering of field data; 5) data analysis and interpretation; 6) description of types; 7) establishing dichotomous keys to classification units; 8) mapping of classification units; and 9) refinement of classification, establishing relative priorities for new data collection. Our approach is qualitative and rule-based, focusing on steps 1 and 2 above. We used existing information from other classifications as much as possible. In particular, we utilized the existing ecoregional frameworks provided by ECOMAP (USDA Forest Service 1999), particularly at the division level, to organize the process of defining systems. We relied on available interpretations of vegetation and ecosystem patterns across the study area. And we reviewed associations of the IVC/NVC in order to help define the limits of systems. Thus our approach draws extensively on the existing literature available to us as well as on the extensive field experience of the contributors.

We divided NatureServe and natural heritage program ecologists into teams, based on Ecological Divisions (Figure 2). Each team worked on developing systems within their division, noting those systems whose range might extend outside the division. After all systems were described, we conducted an overall review of all systems for eastern North America and western North America to ensure consistency of concepts. In recent years we also conducted a number of tests of our systems approach (e.g. Marshall et al. 2000, Moore et al. 2001, Hall et al. 2001, Nachlinger et al. 2001, Neely et al. 2001, Menard and Lauver. 2002, Tuhy et al. 2002, Comer et al. 2002). In particular, we tested how well a systems approach could facilitate mapping of ecological patterns at intermediate scales across the landscape. These tests have led to the rule sets and protocols presented here.

Classification Structure

The structure of the ecological systems classification could be described as "modular" in that it aggregates diagnostic classifiers in multiple, varying combinations. This approach gives us maximum flexibility in the definition of multi-factor units. In addition, we explicitly link our units to two existing hierarchies 1) the vegetation hierarchy of the NVC, which provides a set of units from fine-scaled floristic units to coarse-scaled formation units, and 2) the landscape ecosystem hierarchy of ECOMAP (Bailey 1995, USDA Forest Service 1999), particularly the levels from division down to subsection (see Box 1). For the vegetation hierarchy we emphasize the linkage to association units, and for the landscape hierarchy, we emphasize the Division level. Through database queries, we have also made it possible to link units to the broad-scale map categories used for the National Land Cover Data (Forest, Shrubland, Herbaceous, Woody Wetland, Herbaceous Wetland, Sparse or “barren” etc.).

However, some type of hierarchy for ecological system units may be advantageous. With approximately 600 upland and wetland system types across the lower 48 United States, a hierarchy would at least improve the organization of the units. But, more importantly, a hierarchy may also allow us to further interpret the ecological patterns over a range of intermediate scales. Hierarchical arrangements of biotopes or habitats in Europe (such as by EUNIS) may provide some guidance on establishing a hierarchy of ecological systems presented here.

Development of Diagnostic Criteria and Descriptions

Diagramming factors. Multiple diagnostic criteria may be arranged to allow for a visual expression of the combinations that define each ecological system unit. Figure 3 depicts a subset of ecological system types that are found in the Laurentian – Acadian Division. The major break between “upland” and “wetland” was used as the initial stratifier. Matrix scale physiognomic breaks between “forested” vs. “non-forested were then introduced. Within these classifiers, the primary disturbance regime, topography, climate, and soils were used to further distinguish systems. These finer-scale classifiers set up constraints on the type of floristic patterns that are associated with the systems. This type of diagramming visually displays the logic of how major diagnostic classifiers are organized in developing systems. Subsequent description and qualitative analysis allow these initial assumptions to be tested, then built upon.

Qualitative description. Each type is described in a database that includes a summary of known distribution, environmental setting, vegetation structure and composition, and dynamic processes. A separate portion of the database allows any combination of diagnostic classifiers to be attributed. This permits subsequent sorts and further evaluation of types using any combination of diagnostic classifiers (e.g. all riparian systems, all subalpine systems, all systems found in the Colorado Plateau, etc.).

Attribution of Plant Community Types. NVC associations are used to further describe each unit wherever possible. Vegetation classification units in common usage in both California (Sawyer and Keeler-Wolf 1995) as well as in Alaska (Viereck et al. 1992) were also used when the NVC was incomplete in those areas. Documented associations/communities are listed when there is evidence that they are found in conditions described by the diagnostic criteria. Any occurrence of a given ecological system will have some, but not necessarily all, of the listed communities.

General stratifier	UPLANDS								
Matrix physiognomy	Mostly non-forested	Forested (matrix for this Division)							
Primary disturbance regime		Frequent fire Corollaries: drier soils, more nutrient poor			Windthrow (fires yes but less frequent) Corollaries: more mesic, more nutrients				
Topography		Rolling terrain	Sandplains & coarse outwash		Valley bottoms & extensive flats	Slopes and ridges			
Climate			Temperate	Near-boreal		Temperate	Near-boreal		
Soils							Enriched soils	Acidic	
Floristics								Lowland/ interior	Appalachian
System		**Laurentian-Acadian White Pine - Red Pine Forest**	**Laurentian Pine - Oak Barrens**	**Acadian Near-Boreal Spruce Barrens**	**Acadian Near-Boreal Spruce Flat**	**Laurentian-Acadian Pine-Hemlock - Hardwood Forest**	**Laurentian-Acadian Northern Hardwoods Forest**	**Acadian Lowland Spruce – Fir – Hardwood Forest**	**Acadian Montane Spruce – Fir –Hardwood Forest**

General stratifier	***UPLANDS (continued)***							
Physiognomy	Mostly non-forested							
Elevation (gross)	Montane		Lowland					
Elevation (fine)	Alpine	Subalpine						
Landforms			Rocky hills				flats	
Specialized substrate			Cliff & talus		Rocky outcrops			
Mesoclimate							Inland	Maritime
Chemistry			Acidic	Circumneutral to calcareous	Acidic	Circumneutral to calcareous		
System	**Acadian Alpine Barrens**	**Acadian Subalpine Woodland and Barrens**	**Laurentian-Acadian Acidic Cliff & Talus**	**Laurentian-Acadian Calcareous Cliff & Talus**	**Laurentian-Acadian Acidic Rocky Outcrop**	**Laurentian-Acadian Calcareous Rocky Outcrop**	**Great Lakes Alvar**	**Acadian - North Atlantic Rocky Coast**

Figure 3. Sample decision matrix for classification of selected ecological systems found in the Laurentian-Acadian Ecological Division.

Also, since associations/communities are principally used as descriptors of system units, some could be predicted to occur within more than one ecological system type.

Pattern Type

Review of broad scale ecological pattern for a given region should result in an initial suite of ecological system types that could fall into one of four spatial categories ("matrix, large patch, small patch, linear") (Anderson et al 1999, Poiani et al 2000; see Table 1). For example, matrix-forming forests, shrublands, and/or grasslands may dominate uplands for a given regional landscape. Knowledge of environmental variation, dynamic processes, and resulting compositional variations can be used to qualitatively characterize system types that typically occur in patches ranging from 2,000 on up to 10,000s of hectares. Both large patch and small patch systems tend to appear nested within matrix system types, while linear system types occur along riverine corridors, coastal areas, and major physiographic breaks (e.g., escarpments or cliff faces). Analysis of local-scale patterns nested within a region's natural matrix clarifies the diversity of potential patch and linear system types.

We use these four categories of spatial scale in order to avoid subsuming distinctive biotic and abiotic factors into larger systems, where those factors are clearly different from the matrix or large patch systems. But, the smaller the potential system, the more distinctive these factors needed to be to justify recognizing it as distinct. Thus, e.g., seepage fens are distinguished from their surrounding matrix forests or large-patch floodplain systems because of the distinctive biotic and abiotic factors present, whereas ox-bows or backwater swamps are not distinguished within a floodplain system.

The concepts of both "linear" and "small patch" types typically result in the definition of units that clearly fall into either category. The same is not always true with "large patch" vs. "matrix" types. There are circumstances where an ecological system forms the matrix within one part of its range, but then occurs as a "large patch" type in another part of its range. This likely results in differing dynamics of climate and related disturbance processes – and interactions with other systems – that vary in ways unique to each system type. For example, a savanna system may form the matrix of one ecoregion where landscape-scale fire regimes have historically been supported by regional climate. An adjacent, more humid ecoregion might support the same type of savanna system, but occuring as patches within a matrix of forests. Importantly, we have established as a classification rule that this type of change in spatial character – between "large patch" and "matrix" categories across the range of a type does not force the distinction between two system types. The environmental and disturbance dynamics that result in that variation can be described and addressed for conservation purposes without defining a distinct type.

Table 1. Categories for patch types used to describe ecological systems

Patch Type	Definition
Matrix	Ecological Systems that form extensive and contiguous cover, occur on the most extensive landforms, and typically have wide ecological tolerances. Disturbance patches typically occupy a relatively small percentage (e.g. <5%) of the total occurrence. In undisturbed conditions, typical occurrences range in size from 2,000 to 10,000s ha.
Large Patch	Ecological Systems that form large areas of interrupted cover and typically have narrower ranges of ecological tolerances than matrix types. Individual disturbance events tend to occupy patches that can encompass a large proportion of the overall occurrence (e.g. >20%). Given common disturbance dynamics, these types may tend to shift somewhat in location within large landscapes over time spans of several hundred years. In undisturbed conditions, typical occurrences range from 50-2,000 ha.
Small patch	Ecological Systems that form small, discrete areas of vegetation cover typically limited in distribution by localized environmental features. In undisturbed conditions, typical occurrences range from 1-50 ha.
Linear	Ecological Systems that occur as linear strips. They are often ecotonal between terrestrial and aquatic ecosystems. In undisturbed conditions, typical occurrences range in linear distance from 0.5 to 100 km.

Nomenclature for Ecological Systems

The nomenclature for the ecological systems classification includes three primary components that communicate regional distribution (predominant Ecological Division), vegetation physiognomy and composition, and/or environmental setting. The final name is a combination of these ecological characteristics with consideration given to local usage and practicality.

Ecological Divisions: The Division-scaled units typically form part of each classification unit's name. For example, a "Rocky Mountain" ecological system unit is entirely or predominantly found (>80% of its total range) within the Rocky Mountain Division, but could also occur in neighboring Divisions. This nomenclatural standard is applicable to most ecological system units, except for those types that span many several Divisions (e.g., some tidal or freshwater marsh systems), or that are more localized (>80% of the range) within a subunit of the Division (e.g., Colorado Plateau, within the Inter-Mountain Basins Division).

Vegetation Structure and Composition: Vegetation structure (e.g., Forest and Woodland, Grassland), and vegetation composition (e.g. Pinyon-Juniper, mixed conifer) is commonly used in the name of a system. In sparse to unvegetated types, reference to characteristic landforms (e.g., badland, cliff) may substitute for vegetation structure and/or composition. It will typically come after Ecological Division, but may come before or after *Environment*.

Environment: Environmental factors (e.g., xeric, flats, montane) can be used in conjunction with Vegetation Structure and Composition or, on their own, to name system types. This will typically come after *Ecological Division*, but may come before or after *Vegetation Structure and Composition*.

Examples:
Laurentian-Acadian Pine-Hemlock-Hardwood Forest
Cross Timbers Oak Forest and Woodland
Central Appalachian Limestone Glade and Woodland
Southern and Central Appalachian Cove Forest
North-Central Interior Shrub-Graminoid Alkaline Fen
Cross Timbers Oak Forest and Woodland
Western Great Plains Wooded Draw and Ravine
Rocky Mountain Foothill Grassland
Chihuahuan-Sonoran Desert Bottomland and Swale Grassland

Results

Number and Distribution of Systems

This project identified and described 599 upland and wetland ecological systems within the project area. They represent the full range of natural gradients, with some 381 types (63%) being uplands, 183 types (31%) being wetland, and 35 types (6%) being complexes of uplands and wetlands. Excluding upland/wetland complexes, some 322 types (54%) are predominantly forest, woodland, and/or shrubland, and some 166 types (28%) are predominantly herbaceous, savanna, or shrub steppe. Seventy-four systems (12%) are sparsely vegetated.

A geographic breakdown of ecological system types indicates some expected patterns. Using continental Domain units as one frame of reference (Bailey 1998), within the project area, some 430 types are known to occur in the Humid Temperate Domain (all Pacific coast regions and nearly all of the eastern United States). Another 246 types are attributed to the Dry Domain (from the western Great Plains across the Intermountain West), and 21 units occur in the Humid Tropical Domain (south Florida). Figure 4 indicates the numbers of ecological system units by Ecological Division. The relatively large number of types found in the Gulf and Atlantic Coastal Plain and Central Interior and Appalachian divisions is not unexpected. Each of these large and complex divisions has over 100 ecological system units attributed. Divisions that encompass most of the West, including the Rocky Mountain Division, North American Pacific Maritime, Inter-Mountain Basins, and Mediterranean California include between 60 and 90 types each. The Laurentian-Acadian, Eastern Great Plains, Western Great Plains, and North American Warm Desert divisions each include between 31 and 60 types. Both the Madrean Semidesert and the Caribbean divisions include portions within the coterminous United States, but data from remaining portions were not included in this project area.

Figure 5 depicts numbers of ecological system units within each ecoregion currently used by The Nature Conservancy within the project area. These range from highs of nearly 50 types in the Great Lakes and several Rocky Mountain ecoregions to a low of fewer than 10 for the Mississippi River Alluvial Plain. The mean number for ecoregions included in the project area was 25 types. This obviously varies by size and complexity of the ecoregion.

Figure 6 depicts the number of ecological system units for each state in the coterminous United States. Again, numbers vary by size and ecological complexity of each state. Over 100 units are attributed to Oregon and California. The states of Texas, Virginia, Washington, New Mexico and Arizona include between 70 and 100 types each. Some 13 states, from Michigan to Florida include between 51 and 70 types each. Another 17 states, from Minnesota to South Dakota include between

30 and 50 types. The remaining 11 states in the project area each have fewer than 30 types currently attributed.

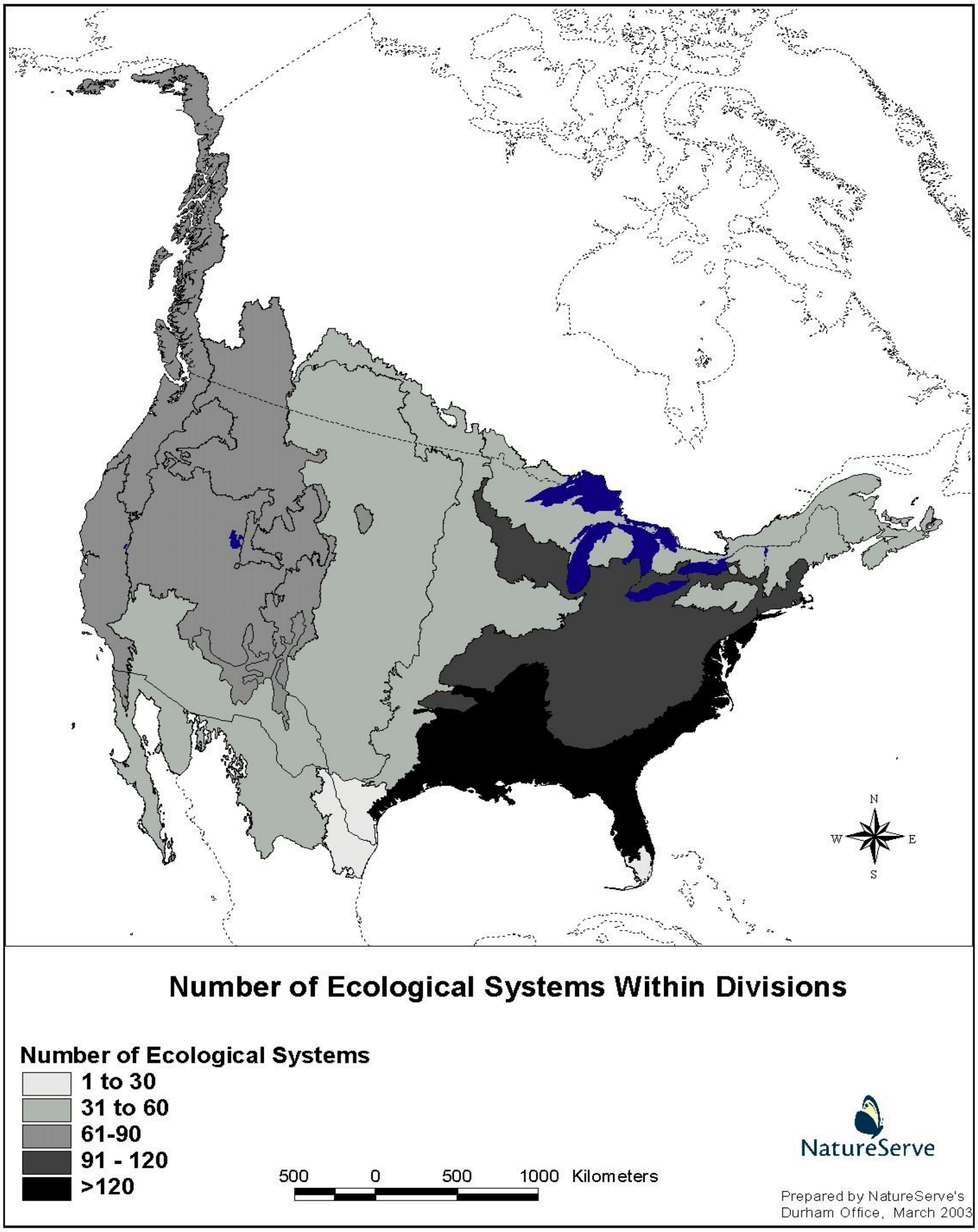


Figure 4. Number of Terrestrial Ecological System types by Ecological Division.

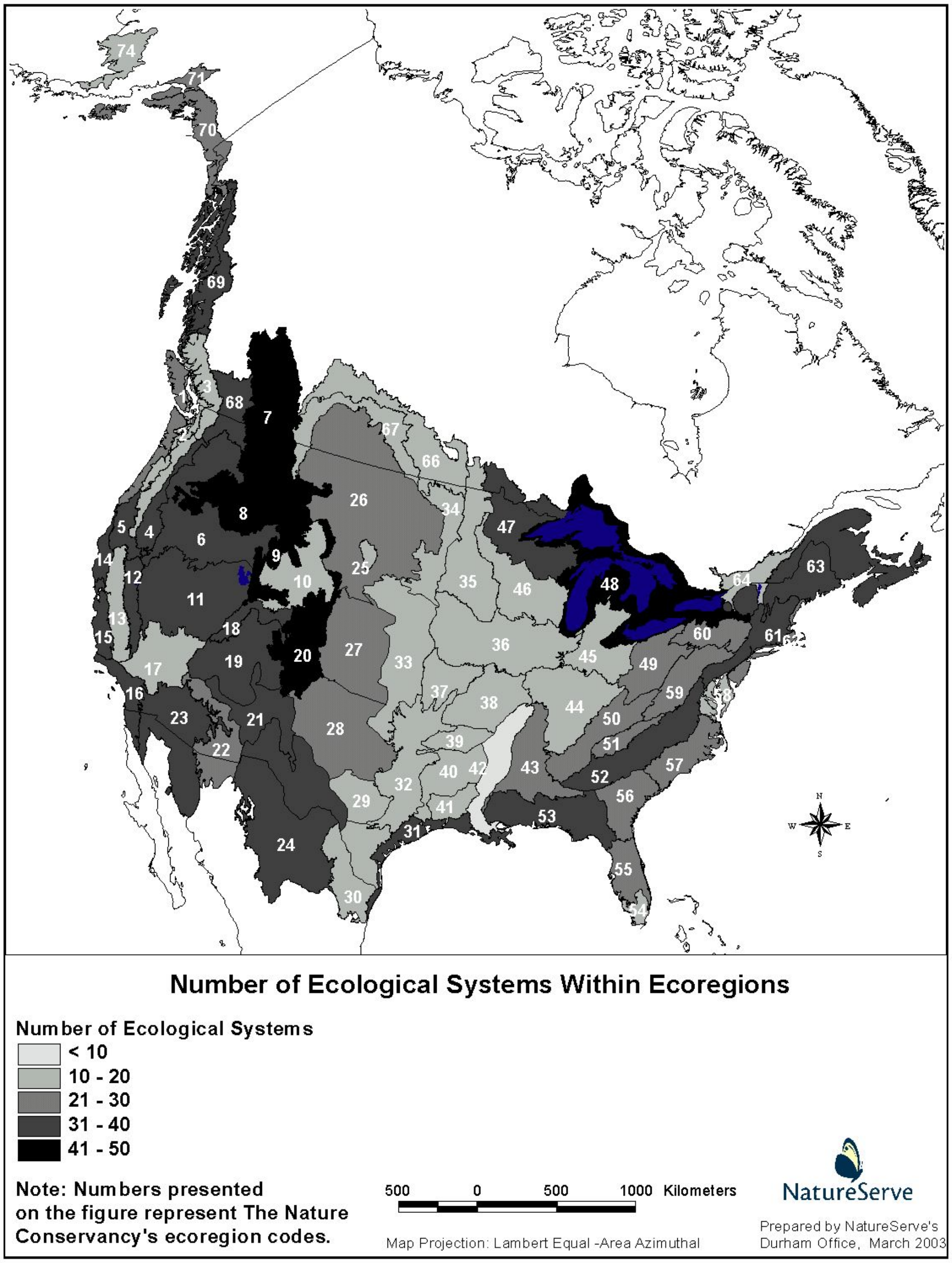


Figure 5. Number of Terrestrial Ecological System types by Ecoregion.

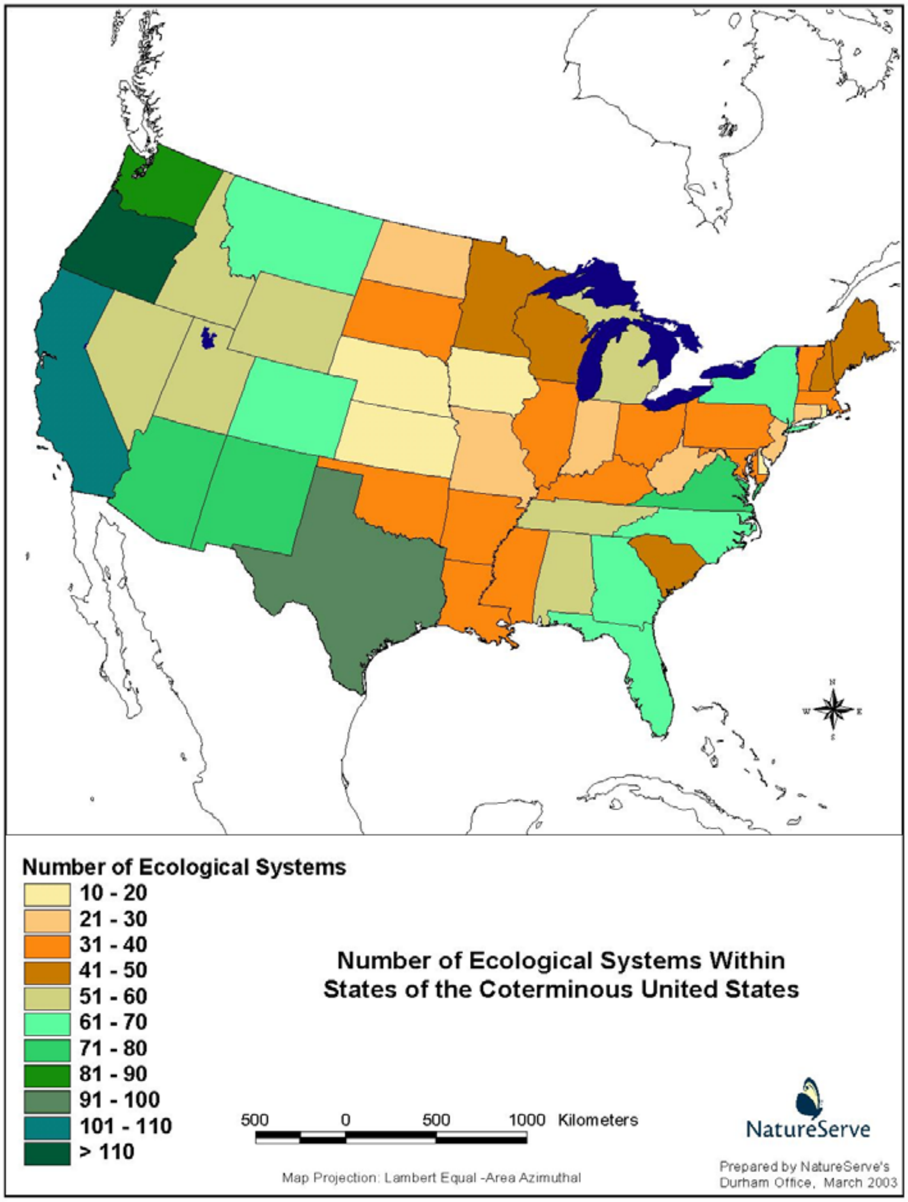


Figure 6. Number of Terrestrial Ecological System types by State.

Linking System Types to Land Cover Types

Table 2 includes a tally of ecological system types and approximations of total area in categories that closely match those used for mapping land cover in the National Land Cover Data (NLCD) managed by the USGS Biological Resources Division. The table also illustrates relative diversity of ecological system types in comparison to total mapped area for the coterminous United States *circa*

1992. In these terms, both herbaceous and woody wetland types, as well as sparely vegetated types are relatively diverse, followed by forests, shrublands, and herbaceous types.

In the NLCD classification, the "Forest" class is a combination of the "Forest" and "Woodland" Formation Classes in the National Vegetation Classification (NVC). Similarly, the NLCD "Shrubland" class encompasses the "Shrubland" and "Dwarf-shrubland" Formation Class of the NVC, and NLCD "Grasslands/Herbaceous" matches the "Herbaceous" Formation Class of the NVC. The NLCD "Woody Herbaceous" class includes upland NVC Formation Groups of "Temperate or subpolar grassland with a sparse tree layer" and "Temperate and subpolar grassland with a sparse shrub layer." This class is not comprehensively mapped in the NLCD. NLCD "Woody Wetlands" encompasses some 80 wetland and saturated Forest, Woodland, and Shrubland Formations of the NVC. Some 43 wetland and saturated Herbaceous NVC Formations make up the "Emergent/ Herbaceous Wetland" class of NLCD. The NLCD "Bare Rock" class closely matches the NVC Sparse Vegetation Formation Class, but could also include areas classified in the Nonvascular Formation class of the NVC.

Table 2. Breakdown of ecological system types in terms of prevailing vegetation physiognomy and upland/wetland status, closely matching categories mapped in National Land Cover Data.

Prevailing Physiognomy and Environment (modified from NLCD 1992)	**Number of Ecological System Types**	**Percentage of Total Number of Types**	**Area in Coterminous United States (*circa* 1992) [miles^2 and %]**
Forest (Evergreen, Deciduous, Mixed)	152	25%	879,858 (29%)
Shrubland (Tall, Short, Dwarf)	71	12%	564,713 (19%)
Woody Herbaceous	30	5%	N/A
Grasslands/Herbaceous	56	9%	479,074 (16%)
Woody Wetlands	100	17%	85,412 (3%)
Emergent/Herbaceous Wetlands	83	14%	37,982 (1%)
Mixed Upland and Wetland	35	6%	N/A
Bare Rock (Sparsely Vegetated)	74	12%	42,640 (1%)

Data Management and Access

The classification information is stored in a MS-Access database (*Systems2000.mdb*). The database includes descriptions of the approximately 600 systems types, their distribution by states and ecoregions, the list of NVC associations that characterize them, and many literature references. It also includes the diagnostic classifiers used to define the ecological systems. Small subsets of systems from several TNC ecoregions also have Element Occurrence (EO) Specifications and EO Rank Specifications stored in the database (see also Appendix 2). The database is available in both Access 97 and Access 2000 versions, in both cases in read-only format. An accompanying manual in

MS-Word (*Systems database manual.doc*) documents its content, functionality, and reporting capabilities.

During 2003, all of the US Terrestrial Ecological Systems and their accompanying data will be converted into NatureServe's central data management system, *Biotics 4*. Once the system types and the data are stored in *Biotics 4*, the full data management, updating, and revision capabilities of that will be available for the continuing development and refinement of system types. In addition, the ecological systems will be served on-line via NatureServe's public website (www.natureserve.org), and NatureServe Explorer, an online searchable databases of species and ecological communities (www.natureserve.org/explorer).

Applications

Applications to Conservation Assessment

Conservation assessment occurs at varying spatial scales to serve the needs of various users. Assessment at a regional scale is often necessary to evaluate status and trends in regional biodiversity. Places are then identified that capture ecological and genetic variation across a broad range of environmental gradients (Johnson et al. 1999). At these regional scales, planning efforts may identify networks of places that, taken together, fully represent characteristic biological diversity. One might then identify areas where more intensive natural resource development could take place in a compatible fashion. That network of places is sometimes referred to as a "portfolio," because a variety of approaches may be used to conserve biological diversity over time through on-the-ground actions. As knowledge expands, and the "market" for conservation changes, one can expect that new places will gain importance, while other places may contribute less to conservation goals. Much like a financial portfolio, a regional conservation portfolio is flexible and priority-based.

Assessments using ecoregions as a spatial planning framework have become increasingly common in recent years, and standardized classifications of ecological systems can serve a central role. Ecoregions are regional landscapes, or relatively large areas of land and water defined by similar geology, landforms, climates and ecological processes. Further, ecoregions contain geographically distinct assemblages of ecological systems that share a large majority of their communities, species, dynamics, and environmental conditions, and function effectively as a framework for conservation at global and continental scales (Bailey 1996, Olsen et al. 2001). In most instances, upland and wetland ecological systems can be mapped comprehensively across ecoregions or any other regional planning area. Therefore they aid in evaluating the status and trends of numerous ecological phenomena, from trends in land conversion or wildlife habitats to creating repeatable metrics for landscape fragmentation. Because ecological system units are defined to represent characteristic composition, structure, and function at intermediate scales, conservation goals aimed at conserving ecological systems should also capture ecological processes important to many, but not all, biological communities and species.

An "element-based" approach to conservation assessment commonly establishes a suite of species, communities, and ecological systems that provide the focus for representing biodiversity. An additional suite of elements may also be included in the analysis to represent overall conservation value (e.g., those identified under environmental regulations, open space, scenic or cultural values.). The objective should be to select a limited set of elements that could serve as effective surrogates for all (or nearly all) biological diversity. Through conservation of these elements across the planning area, one seeks to efficiently secure the ecological environments and dynamic interactions that support the vast majority of species. Occurrences of these elements, as well as the relative quality of their occurrences, are used to characterize biodiversity value and identify specific locations for conservation action.

To identify these elements effectively, one may use several sets of selection criteria. Typically one should include elements from multiple levels of ecological organization, elements representing varying degrees of rarity, vulnerability, and endemism (Appendix 3), and elements representing multiple geographic scales of habitat/area requirement. The outline in Table 3 summarizes recommended criteria to select elements. Elements of biological diversity – the ecological systems, communities, species assemblages, and species — that meet at least one of the criteria in the outline are therefore placed on the list of **selected elements**.

Table 3. Core Selection Criteria for Elements for Biodiversity Conservation

I. **Ecological systems.**
   A. **All** natural/semi-natural terrestrial ecological systems that are known to occur in the planning area.
   B. **All** natural/semi-natural aquatic ecological systems that are known to occur in the planning area.

II. **Ecological communities.**
   A. Rare natural/semi-natural terrestrial plant associations globally ranked G1-G3 by the Natural Heritage Network.
   B. Rare natural/semi-natural aquatic macrohabitats globally ranked G1-G3 (where available).
   C. Vulnerable species assemblages – e.g. areas where concentrations of migratory species occur.

III. **Species** (including infraspecific taxa).
   A. Species globally ranked G1-G3; subspecies/varieties globally ranked T1-T3.
   B. Species (subspecies) globally ranked G4-G5 (T4-T5), that on the whole are "of concern" by virtue of:
      1. Experiencing significant **decline** across their range.
      2. Are currently stable, but **vulnerable to future declines,** due for example to their broad regional landscape requirements or to their concentration in particular areas during their migrations.
      3. Are considered **endemic** to the planning area.
      4. Having **widely disjunct** occurrences in the planning area.
      5. Are considered to be **"keystone"** species.

Using these selection criteria, three levels of biological or ecological organization: *ecological systems*, *communities,* and *species*, are represented among selected elements. As these categories indicate, this reflects a "coarse filter/fine filter" hypothesis – i.e. the conservation of multiple, high-quality occurrences of all ecological systems will also support the majority of native biodiversity. Since this "coarse filter" on its own would be unlikely to represent all biodiversity, especially those that are rare and thus not reliably found within most examples of ecological systems, additional elements, those that are imperiled or vulnerable, are also needed – the “fine filter.” Experience suggests that this is the most efficient and effective approach to capturing biodiversity in a network of reserves (e.g. Jenkins 1976, 1985; Noss and Cooperider 1994, Haufler et al. 1996, Groves et al. 2002, Kintsch and Urban 2002). The coarse filter/fine filter approach also reduces complexity and cost associated with strict species-based approaches (e.g. Scott et al 1987, Beissinger and Westphal 1998; Willis and Whittaker 2002) while allowing sufficient flexibility to integrate new approaches as technical hurdles are overcome (e.g. Fleishman et al. 2001, Carroll et al. 2001, Scott et al. 2002).

Careful element selection therefore provides appropriate focus for efforts to map and evaluate element occurrences, then establish specific conservation goals and objectives.

Applications to Element Occurrence Inventory and Mapping

Element Occurrences: Information on status and trends of ecosystems is critical for evaluation, conservation, and management of natural resources. NatureServe and natural heritage scientists develop detailed information about the location and viability or integrity of biodiversity elements and about the sites that are important for their persistence or survival. They help reduce negative impacts on biodiversity by providing this information in ways that facilitate awareness of the key impacts that various development projects may have (Stein and Davis 2000). Here we discuss the first key part of the mission as it relates to ecological systems - identifying the systems on the ground and developing detailed information on their locations or occurrences (“element occurrence specifications”). In the next section (Applications to Management and Monitoring), we introduce the issue of assessing the ecological integrity of these occurrences (see also Appendix 2).

Elements, then are the units of biodiversity, whether species, communities, or systems. Element occurrences are geographic locations of those elements on the ground. Specifically, NatureServe standards (NatureServe 2002) state that:

> *An element occurrences (EO) is an area of land and/or water in which a species, natural community, or ecological system is, or was, present. An EO should have practical conservation value for the Element…. For community Elements, the EO may represent a stand or patch of a natural community, or a cluster of stands or patches of a natural community. For system elements, the EO may represent a cluster of stands from different communities that are part of the system.*

Element occurrences are the principal source of information about the distribution of the elements. The occurrences are typically mapped, often at the scale of 1:24,000, but scale can vary depending on the application.

Key to the identification and mapping process is establishing the specifications for a given occurrence. When is one occurrence of a system distinct from another occurrence of the same system? For example, a hemlock-hardwood system (such as the Appalachian Hemlock-Hardwood Forest) may occupy a series of ravines, particularly on cooler north slopes, distinct from either the riparian forests in the bottoms of ravines or oak forests that predominate on the warmer and drier upland slopes. How far apart do the hemlock stands need to be before they are treated as separate occurrences? And do small hemlock stands of only 0.5 hectares get recorded as a separate occurrence from the oak systems that surround it? It is these questions about minimum patch size and separation distances between patches that are addressed by the "element occurrence specifications" (EOSPECS), which ensure consistent application of the systems approach.

Defining EOs. For ecological systems (as for communities), EOs represent a defined area that contains (or contained) a characteristic ecological setting and vegetation. EOs are separated from each other by barriers to species interactions or ecological processes, or by specific distances defined for each element across adjacent areas occupied by other natural or semi-natural community types, or by cultural vegetation. EOs can be created for both communities and systems. In some cases a system EO may encompass several community-level EOs, either of the same community type (in cases where the separation distance requirement at the systems level is greater than at the community level) or several community types.

Recommended minimum sizes for the system types will meet or exceed those of the component community types.

They are:
- 10 ha for matrix,
- 10 ha for upland large patch;
- 1 ha for wetland large patch;
- 0.5 ha for small patch;
- 100 m for all linear types.

Stands/areas below the recommended minimum size become difficult to judge in terms of community or system type characteristics, and, if isolated, become heavily influenced by edge effects. For conservation purposes, generally only larger sized occurrences of each community or system type are tracked and the threshold for minimum size is seldom approached.

Barriers and Separation Distances. Known barriers for Elements, either naturally occurring or manmade, should be described in the EO specifications. For community or system EOs, barriers may be obstacles that limit the expansion or alter the function of these types. These barriers either separate populations of most of the component species within the community or system, thus obstructing or severely limiting gene flow and ecological interactions, or they obstruct or limit ecological processes that these species depend on. Barriers may be common for many wetland communities or systems, but are typically less common for many upland terrestrial communities or systems.

In addition to barriers that totally, or almost completely, prevent ecological processes and species interactions, there may be habitats between two stands of an element that partially restrict species interactions or ecological processes. Unlike barriers, their effect depends on the kind and extent of this intervening habitat. This leads to the issue of separation distance. Assigning values for separation distances between two stands promotes consistency in the manner in which EOs are defined and mapped. Smaller separation distances are used when the intervening habitat is highly restrictive to the ecological processes or species interactions the element depends, and greater distances are used when these habitats are less prohibitive to ecological processes or species interactions.

We use two broad categories of intervening habitats to define separation distances, namely – natural/semi-natural vegetation or cultural vegetation. Generally speaking, intervening natural and semi-natural vegetation will have less of an ecological effect between two stands of an EO than intervening cultural vegetation. Thus rather simplistically, we suggest that different separation distances be specified for these two kinds of situations. Typically, a shorter separation distance is specified when the intervening habitat is cultural vegetation than when it is natural/semi-natural. Minimum values for separation distances have been recommended to ensure that EOs are not separated by unreasonably small distances, which would lead to the identification of unnecessarily splintered stands as potential targets for conservation planning or action. For communities or systems, the minimum separation distance for intervening areas of different natural or semi-natural communities is set at 1 km or greater, and for intervening areas of cultural vegetation, the distance is set at 0.5 km or greater (Table 4). These separation distances may, of course, be much larger. For communities or systems found primarily in mountainous regions, where habitat tends to be less fragmented, separation distances may be 5 km or more. A few elements may require separation distances that are less than the established minimum; in such cases, these distances should be justified in the EO specifications. Again, more detailed explanation and examples of these issues are found in Appendix 2.

These separation distances may be further refined by considering the kind of natural/semi-natural or cultural vegetation present. Intervening natural and semi-natural areas with similar kinds of habitat characteristics to the stands of a community or system under consideration will have less of an effect on community or system processes than those with very different kinds of characteristics. For example, bog stands separated by intervening areas of upland jack pine on bedrock could be more readily treated as distinct EOs than bogs separated by areas of black spruce swamp. However, at this time, no specific guidelines are suggested for these situations.

Table 4. Recommended Minimum Separation Distances for Communities and Ecological Systems

Type of Separation	Minimum Separation Distance
Barrier	qualitatively defined
cultural vegetation	≥ 0.5 km
different natural or semi-natural communities or systems	≥ 1 km

Applications to Comprehensive Mapping

Comprehensive mapping of terrestrial ecological systems draws heavily on the experience of mapping vegetation using remotely sensed imagery and ancillary data (e.g., the USGS-BRD/NPS Vegetation Mapping Program standards as outlined by Grossman *et al.* 1994; Faber-Langendoen *et al.* 2002). That methodology recognizes that vegetation forms one of the most readily observable natural features of the landscape. It provides an important measure of the current condition of natural systems and can serve as a cost-effective monitoring tool for ongoing management of those systems. Vegetation mapping is the process of integrating multiple sets of information. It often involves interpreting signatures from vegetation from remotely sensed data – sometimes integrating ancillary spatial data - then assigning each signature to a map unit. In order to ensure that each mapper bases his or her interpretation of those signatures on the same ecological perspective, a consistent classification is needed.

Given the inherent difficulties in achieving a consistent classification scheme, it may appear that classification should really be the end result of mapping; that is, the vegetation mapper is free to explore the vegetation patterns as they appear on the local landscape, and choose those features that are most relevant to the species combinations and environmental factors on hand (*a posteriori* classification). Indeed, Kuchler (1988) argued that this approach has much to recommend it. But Kuchler also pointed out that such *a posteriori* classifications have a major drawback – they are best applicable only in the mapped area or, at best, only short distances beyond the borders of the area. Since the scope of both the National Vegetation Classification and the NatureServe Ecological Systems Classification is national - indeed hemispheric - basing the mapping on these classifications should allow any map produced to be compared to other areas throughout the country in an integrated and consistent manner. It is for that reason, for example, that federal agencies such as the USGS Gap Analysis Program and the National Park Service chose in the mid-1990s to work with an *a priori* classification, the NVC, seeking to balance the needs of mapping local vegetation patterns with the overall need to achieve consistency across the nation.

Mapping Issues with the NVC: The stated intention of the Gap Analysis Program for land cover mapping has been to depict vegetation matching the scale and concept of the vegetation Alliance, as described in the NVC. However, not all vegetation types are equally mappable at a given geographic scale. GAP efforts to map vegetation on a statewide scale have had difficulty achieving desired levels of mapping accuracy for map units reflecting all vegetation alliances. This is due to the reality that not all Alliances occur in large and distinctive patches that are easily depicted with satellite imagery. As examples, many wetlands and herbaceous uplands may include several Alliances co-mingled within a few hectares. As one works at scales of multiple states, the problem of consistent Alliance-scale mapping increases. Figure 7 depicts a combined coverage from five central United States (CO, KS, NE, SD, and WY). While Alliance-level units were mapped in each state, the success at achieving this scale varied significantly. In addition, where some states were able to achieve Alliance scale units, their neighboring states that also include the same vegetation types may not have been as successful. As a result, any regional coverage will tend to include fewer Alliance-scale units depicted consistently across the map area than for any given state or subregion. In this instance, only 17 Alliances were mapped consistently across this area; just a small subset of those that are known to exist on the ground.

So while many Alliances can be mapped by using both remotely sensed imagery and an understanding of the ecological factors that help define them (e.g., elevation, soil type, aspect), some Alliances remain indistinguishable using remotely sensed imagery. The reasons for this vary but common examples are that species that differentiate similar Alliances occur beneath a dense canopy of trees or shrubs, that differential species had very similar signatures when the imagery was acquired, or that the scale of the Alliances is below the standard minimum mapping unit.

To maintain the *a priori* classification, the mapping team may consider using higher levels of the NVC hierarchy as map units. NVC units at “middle-levels” of the hierarchy, such as Formation, are driven primarily by vegetation physionomy, rather than considerations of spatial scales and ecological variables. Whereas the NVC Association unit is typically mappable at scales of around 1:24,000 or larger and often corresponds to ecological factors at that scale, it is more difficult to identify typical

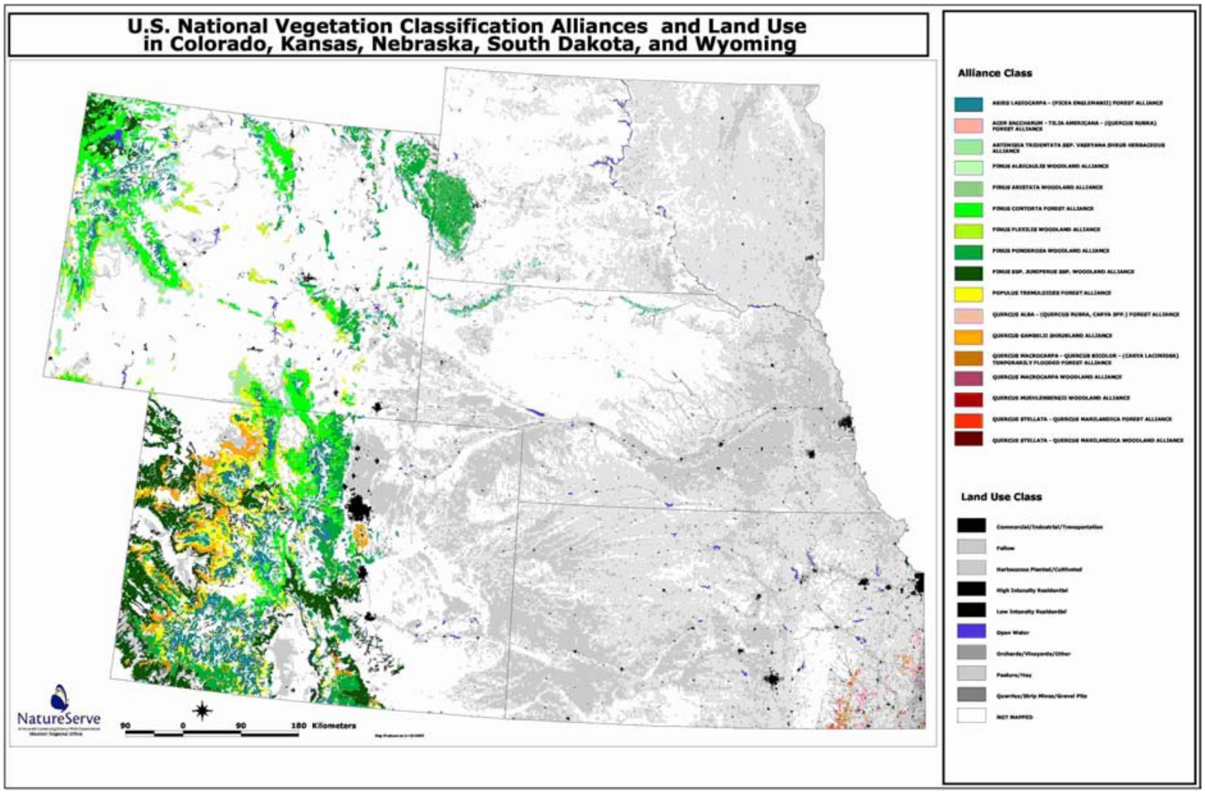


Figure 7. Alliance-scale units mapped comprehensively across CO, KS, NE, SD, and WY (from Comer et al. 2003).

spatial scales and ecological patterns for the mid-level units. So the higher levels of the NVC hierarchy do not necessarily provide suitable classification units for mapping at “coarser” (smaller) scales. Of particular note for applying the NVC to mapping, three aspects are worthy of further exploration: 1) the practical “constraints” imposed by the physiognomic hierarchy on classification units, 2) the variable, and sometime wide, ecological “distance” between Formation, Alliance, and Association levels of the NVC, and 3) potential difficulties for mapping some environmental attributes of the NVC, regardless of minimum map units size.

1. Because the NVC is a strictly nested hierarchy, classification attributes from higher levels are carried over to units further down. So for example, physiognomic distinctions (e.g. forest vs. woodland, evergreen vs. deciduous, needleleaf vs. broadleaf) that enter in the classification at the Class, Subclass, Formation Group, and Formation levels are carried over directly to nested Alliance and Association units. Vegetation types that differ in any one physiognomic attribute

(e.g. forest vs. woodland) form distinct Alliances and Associations, although they may co-mingle on a given landscape.

2. For some types of vegetation, the differences between Formation, Alliance, and Association scales are quite large. For example, a "short bunch temperate or sub-polar grassland Formation" or "lowland or submontane cold deciduous forest Formation" unit likely encompasses hundreds of alliances and thousands of associations around the globe. On the other hand, the "caespitose needle-leaved or microphyllous evergreen dwarf shrubland Formation" or the "creeping or matted drought-deciduous dwarf shrubland Formation" likely include relatively few alliances and associations around the globe. Similarly, some widely distributed Alliances (e.g. *Pinus ponderosa* Woodland Alliance) include much variability, as expressed by over 50 Associations, while other alliances may include only one or just a few associations. This variability among different NVC units can make systematic "aggregations" of map units up from Associations, to Alliances and Formations awkward and often undesirable.
3. Although the NVC hierarchy is primarily based on vegetation, it also uses climatic, topographic and other criteria as a practical tool for dividing the vegetation units. Several environmental attributes enter the NVC hierarchy at the Formation level. Among these are hydrologic modifiers (e.g. temporarily flooded, seasonally flooded, semi-permanently flooded, etc.) that require very detailed, if not multi-temporal, data to accurately apply. So simply "aggregating up" from finer scales to what is often viewed as a rather "coarse" Formation scale still may not solve the mapping problem.

To these considerations we must add the reality of incomplete development for the NVC. Remarkable progress on the classification has been made in the years since 1994. Large portions of some 5,000 Associations have been described; however, parts of the landscape remain inadequately accounted for in the NVC. It is safe to say that we will be coping with our ignorance for some time to come, so the ability to work flexibly at multiple, systematically defined levels of thematic resolution remains highly desirable.

The NVC, therefore, provides a hierarchical classification structure that allows for varying levels of floristic and physiognomic detail, but depending on the circumstances, mapping protocols can easily permit designations of mapping mosaics that are "ad-hoc" or overly driven by observed patterns in available imagery. This, in part, defeats the purpose of an *a priori* classification that is intended to guide the mapping process. One approach to address this situation is to develop classifications above the NVC Association scale that circumvent some of the mapping-related problems inherent in the NVC hierarchy, but still provide units that are practical and useful for management and conservation. Some of the issues identified above could be resolved by revising the NVC hierarchy itself—indeed, the FGDC hierarchy revisions working group proposes to undertake such revisions in the near future. Others, however, require a different approach that focuses on the ecological and spatial relations among the types, rather than just the vegetation relations. The ecological systems classification is intended in part to address this situation.

Ecological Systems provide "meso-scaled" units as a basis for analyzing vegetation patterns, habitat usage by animals and plants, and systems-level comparisons across multiple jurisdictions. They also provide useful, systematically defined, groupings of NVC Alliances and Associations, forming the basis of map units where Alliance and/or Association level mapping is impractical. Figure 8 depicts some 63 terrestrial ecological system units mapped across the same five

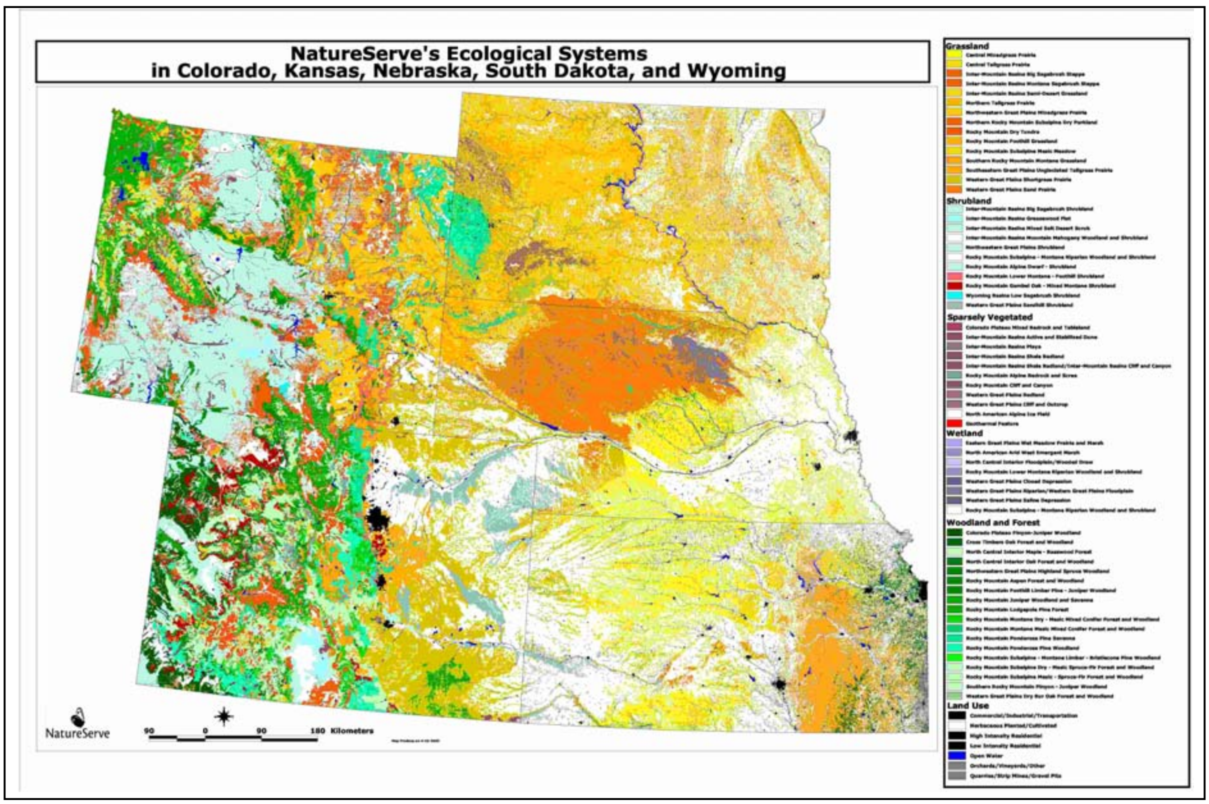


Figure 8. Terrestrial ecological system-scale units mapped comprehensively across CO, KS, NE, SD, and WY (from Comer et al. 2003).

states shown in Figure 7. The same vegetation coverages used for the alliance-level map in Figure 7 were used again, but, in addition, biophysical variables such as elevation, landform, surface geology, soils, and hydrography were also brought in. These variables were combined with the concept statements of each ecological system type to create a map of terrestrial ecological systems (Comer et al. 2003). Not all of the 90 terrestrial ecological system units thought to occur in this five-state region were depicted in this map with existing data. Those not depicted tend to occur as very small patches (e.g. montane fens), or are known to occur primarily in adjacent states, but likely have limited occurrence within this map project area. However, future efforts should have considerably more success if these *a priori* ecological system units are the mapping objective.

Many of the same mapping issues from regional efforts extend to more localized projects, even those where low-elevation aerial photo interpretation is the principal remotely sensed-data. An example from Zion National Park illustrates a common circumstance with more local-scale mapping efforts (Cogan et al. 2002). Here, as with all National Park Service vegetation mapping, the stated *a priori* classification and mapping objective is the NVC Association.

Zion NP is a relatively large park (593 km^2 or 229 mi^2). Major regional floras influence the vegetation, with Mojavean elements in the southwestern portion, Great Basin floristics in the western portion, and influences of the Colorado Plateau and southwestern Utah flora in the eastern and northern portion. Vegetation diversity is high because the elevation gradient extends for nearly a mile (1125-2600 m, 3680-8726 ft) and the landscape is complex. Field-based sampling and classification work in Zion NP resulted in 97 described NVC Associations. Of the 42 natural/near-natural vegetation map units, 20 match the scale and concept of NVC Association, 14 match the NVC Alliance, four match NatureServe Terrestrial Ecological System units, and four would represent a combination of Ecological System units. The 42 original map units correspond to 20 Ecological Systems, providing a park-wide perspective on the Ecological System units found within the park (Figure 8). Figure 8 provides a park-wide perspective on the Ecological System units found within the park and the one-mile buffer, along with aquatic and land use features. Two system types, Great Basin Pinyon-Juniper Woodland and Colorado Plateau Pinyon-Juniper Woodland, were not distinguished in the fine-scale map units, so they are represented as one combined unit. Given our knowledge of the elevation ranges that distinguish these two pinyon-juniper units, they could be feasibly mapped as separate units.

The most prevalent systems across this park landscape include these two types of Pinyon – Juniper Woodlands, Rocky Mountain Gambel Oak Mixed – Montane Shrublands, Colorado Plateau Mixed Bedrock Canyon and Tableland, Rocky Mountain Ponderosa Pine Woodland, and Rocky Mountains Bigtooth Maple Ravine Woodland.

Mapping efforts at Zion NP were ongoing at the time of this publication, but existing data were sufficient for a preliminary accuracy assessment. Raw accuracy scores for each fine-scale map unit yielded a total accuracy of 489 correct points out of 781 samples, or 63% accuracy. A comparable assessment for the map of ecological system units yielded 609 correct points out of 800 samples, or 76% accuracy (Comer et al. 2002).

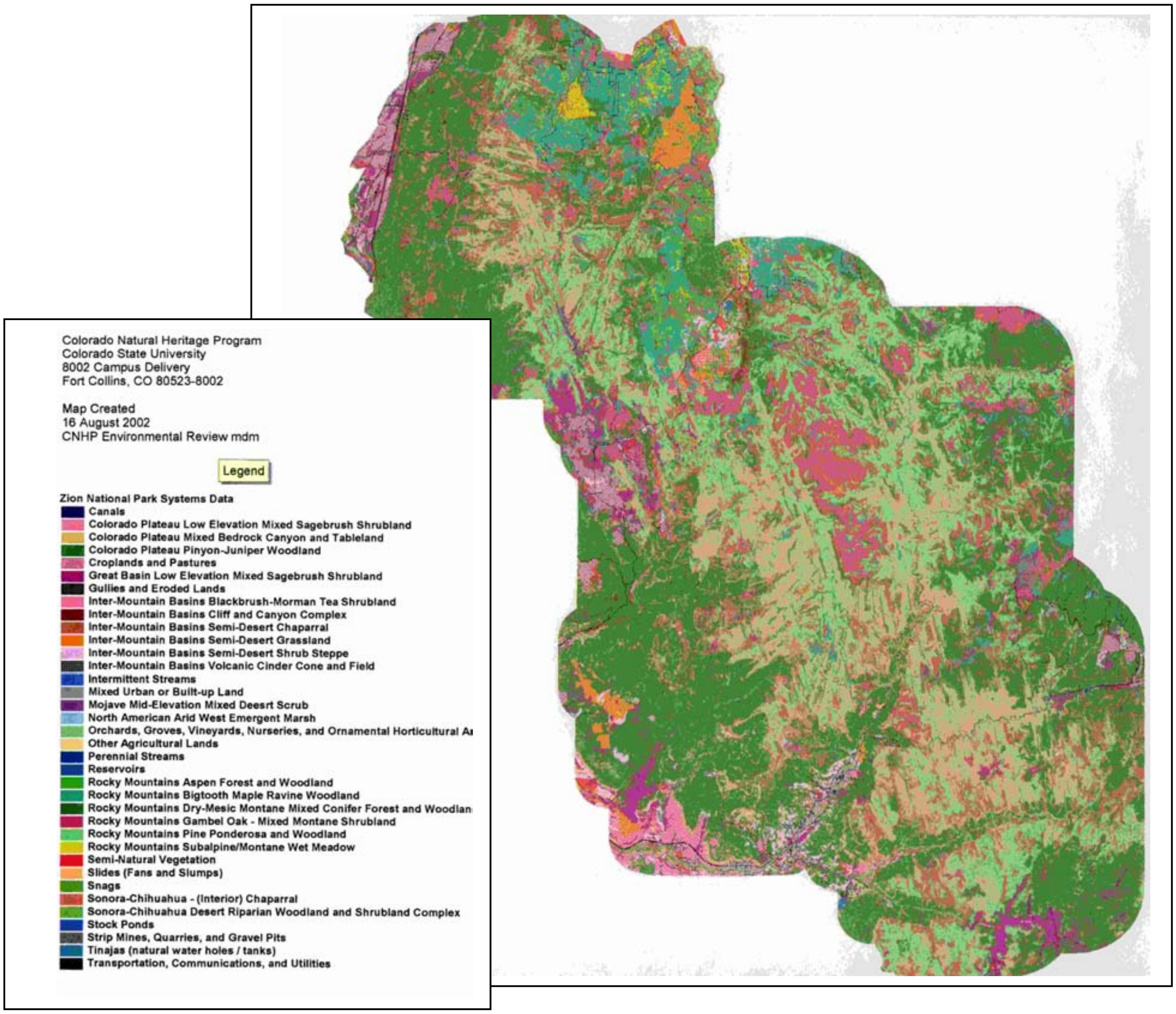


Figure 9. Terrestrial ecological systems of Zion National Park and environs (scale ~ 1: 200,000) (from Comer et al. 2002)

In this and many other examples, mapping ecological system units could provide an additional standard layer of high accuracy. Some of the detail in vegetation structure and composition are lost at the systems scale. For example, at Zion NP, the significant presence of either Gambel oak or big sagebrush in the understory of pinyon – juniper woodland is subsumed into the more broadly defined Colorado Plateau Pinyon – Juniper Woodland unit. Similarly, the understory components of manzanitas and Gambel oak with ponderosa pine is lost, as is the differentiation of sparsely vegetated types dominated by ponderosa pine vs. mountain mahogany vs. the bedrock formations of Carmel Limestone, Navajo Sandstone, and Temple Cap sandstone. However, for several types, mapping at the ecological systems level would have resulted in the same level of thematic detail as the fine-scale map.

The level of systematic aggregation of Associations represented by Ecological System units presents a number of trade-offs. As noted in the preceding examples, some elements of structure and composition are clearly lost by using Ecological Systems instead of Associations or Alliances. If however, classification and mapping were approached from a multi-scaled perspective, there could be some clear advantages. For example, National Parks could be comprehensively *classified* to the Association level, following current data collection and analysis practices, but then *mapped* using both Ecological System units (comprehensively) and individual Associations or Alliances (where desirable and feasible). Ecological Systems would serve as the default map units, but resource managers would specify those areas or types that should be mapped at the Association level. Similarly, polygons mapped to Ecological System units would continue to have additional layers of detail with other kinds of information that address management purposes. For example, polygons labeled with Ecological System units would still have structural modifiers, such as canopy density and height, even where Association-scale thematic resolution is not feasible.

In summary, highly complex landscape features make high-resolution vegetation mapping through remote sensing extremely difficult. Because Ecological System units integrate the environmental setting into their definition, they lend themselves well to using ancillary data, such as high-resolution digital elevation, hydrography, and soils to “constrain” the options for image processing and reflect important ecological attributes that are provided by remotely sensed data. In most cases, multiple ancillary data sets could be combined with plot data and, with quantitative techniques such as regression trees (e.g. see Hansen et al. 1996; De’ath and Fabricius 2000), one could clarify recurring relationships to provide repeatable decision rules for mapping.

Applications to Management and Monitoring

Having mapped ecological systems and established occurrences on the ground, we want to know if each mapped occurrence is of sufficient quality (viability or ecological integrity) or can feasibly be restored to such quality. This is the next essential step towards developing local-area management and monitoring objectives. Characterizing and evaluating the quality of an occurrence provides the basis for assessing ecological stresses—the degradation, or impairment—of element occurrences at a given site. There are three core components of occurrence evaluation that can be applied to *all focal conservation elements* in a conservation site of any scale – whether these are individual populations or species, assemblages of species, ecological communities, or ecological systems. These core components and their function are as follows:

1) **Key Ecological Attributes** – structure, composition, interactions and abiotic and biotic processes that enable the Element Occurrence to persist.

2) **Indicator** – measurable entity that is used to assess the status and trend of a Key Ecological Attribute.

3) **Indicator rating** – the point within a given expected range of variation one would rate each Indicator that describes its current status.

To assess the quality of element occurrences, one must first identify and document a limited number of key ecological attributes that support them (the terms "key ecological attribute" and "indicators" are comparable to the term "ecological attributes" and "indicator" used by TNC in Parrish et al. 2003 and by the EPA publication of Young and Sanzone 2002). After these are identified, a set of measurable indicators are established to evaluate each attribute and document their expected ranges of variation. For each indicator, we may then establish thresholds for distinguishing their current status along a relative scale from "Excellent" to "Poor."

Documentation of these basic assumptions about key ecological attributes, ranges of variation, thresholds, and indicators for measurement, are called "Element Occurrence Rank Specifications;" and form a central component of Heritage methodology. These specifications allow one to consistently assess whether the attributes exhibited by a given occurrence are within desired ranges or whether they will require significant effort to be maintained or restored to their desired status. Each key attribute is reviewed, rated, and then combined with others to rank each occurrence as A (excellent), B (good), C (fair), and D (poor). The higher the estimated viability or integrity of the occurrence, the higher is its EO rank and presumed conservation value. Table 5 lists the basic EO Ranks assigned to each occurrence. The break between C and D establishes a minimum quality threshold for occurrences. D-ranked occurrences are typically presumed to be beyond practical consideration for ecological restoration. In subsequent management planning, these ranks and underlying attributes and indicators aid in focusing conservation activities and measuring progress toward the local conservation objectives.

Table 5. Basic Element Occurrence Ranks

EO Rank	Description of Ecological Integrity
A	excellent
B	good
C	fair
D	poor
E	verified extant (integrity not assessed)
H	historical (not recently located)
X	extirpated (no longer extant)

Because EO ranks are used to represent the relative conservation value of an EO as it currently exists, EO ranks are based solely on attributes that reflect the present status, or quality, of that EO. The three generalized EO rank categories used to organize the various key ecological attributes are condition, size, and landscape context. Ranks in each of these categories are combined to arrive at an overall occurrence rank. Thus:

$$\text{Condition + Size + Landscape Context} \Rightarrow \text{Estimated Viability or Integrity} \approx \text{EO Rank}$$

For community and system Elements, the term “ecological integrity” is preferable to that of viability (used for species), since communities and systems are comprised of many separate species, each with their own viability. Ecological integrity is the “maintenance of…structure, species composition, and the rate of ecological processes and functions within the bounds of normal disturbance regimes[3]. More directly, EO ranks reflect the degree of negative anthropogenic impact to a community or

[3]From. Lindenmayer and Recher (in Lindenmayer and Franklin 2002). Similarly, Karr and Chu (1995) define ecological (or biological) integrity as “the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, and assemblages) and processes (mutations, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat of a region.

system (*i.e.*, the degree to which people have directly or indirectly adversely or favorably impacted community composition, structure, and/or function, including alteration of natural disturbance processes).

It is not necessary to have knowledge of all factors in each of the three rank categories to develop EO rank specifications. The three EO rank factor categories and generalized key attributes are summarized in Table 6 below.

Table 6. Element Occurrence Rank Categories and Key Ecological Attributes

CATEGORY	GENERALIZED KEY ECOLOGICAL ATTRIBUTES (examples of indicators are noted within parentheses)	Species	Communities and Systems
Condition	reproduction and health (evidence of regular, successful reproduction; age distribution for long-lived species; persistence of clones; vigor, evidence of disease affecting reproduction/survival)	√	
	development/maturity (stability, presence of old-growth)		√
	species composition and biological structure (richness, evenness of species distribution, presence of exotics)	√	√
	ecological processes (degree of disturbance by logging, grazing; changes in hydrology or natural fire regime)	√	√
	abiotic physical/chemical attributes (stability of substrate, physical structure, water quality) [excluding processes]	√	√
Size	area of occupancy	√	√
	population abundance	√	
	population density	√	
	population fluctuation (average population and minimum population in worst foreseeable year)	√	
Landscape Context	landscape structure and extent (pattern, connectivity, *e.g.*, measure of fragmentation/ patchiness, measure of genetic connectivity)	√	√
	condition of the surrounding landscape (*i.e.*, development/maturity, species composition and biological structure, ecological processes, abiotic physical/chemical attributes)	√	√

Indicators. Key Ecological Attributes may be difficult or impossible to directly measure. Where this is the case, an indicator of the Attribute that may be reasonably and effectively measured should be identified. In a river floodplain system, for example, river flow dynamics may be an ecological process tha is a Key Ecological Attribute, but it is not reasonable to expect that every possible parameter would be measured. A few parameters (e.g., flood seasonality and periodicity) can be selected that will give us an overall indication (indicator) of how the status of our Key Attribute (flow dynamics) is changing. So the indicator may be a subset of the variables defining the Key Attribute, or a more measurable substitute for the Attribute.

Any element's Key Ecological Attributes (and therefore their indicators) will vary over time in a relatively undisturbed setting. This variation is not random, but falls within a range that we recognize as either a) natural and consistent with the long-term persistence of each occurrence, or b) outside the natural range because of human influences (e.g., fire suppression in fire adapted systems).

Establishing Thresholds. To effectively evaluate occurrences relative to each other, overall ecological integrity ranks should establish a scale for distinguishing between "A", "B", "C", and "D" occurrences. This scale should usually spread from a lowermost limit (the "D" rank or minimum EO threshold) up through the threshold for an "A" rank. In addition, the threshold delineating EOs with "fair" vs. "poor" viability or integrity must be identified. Figure 10 illustrates the rank scale for "A", "B", "C", and "D"-ranked EOs.

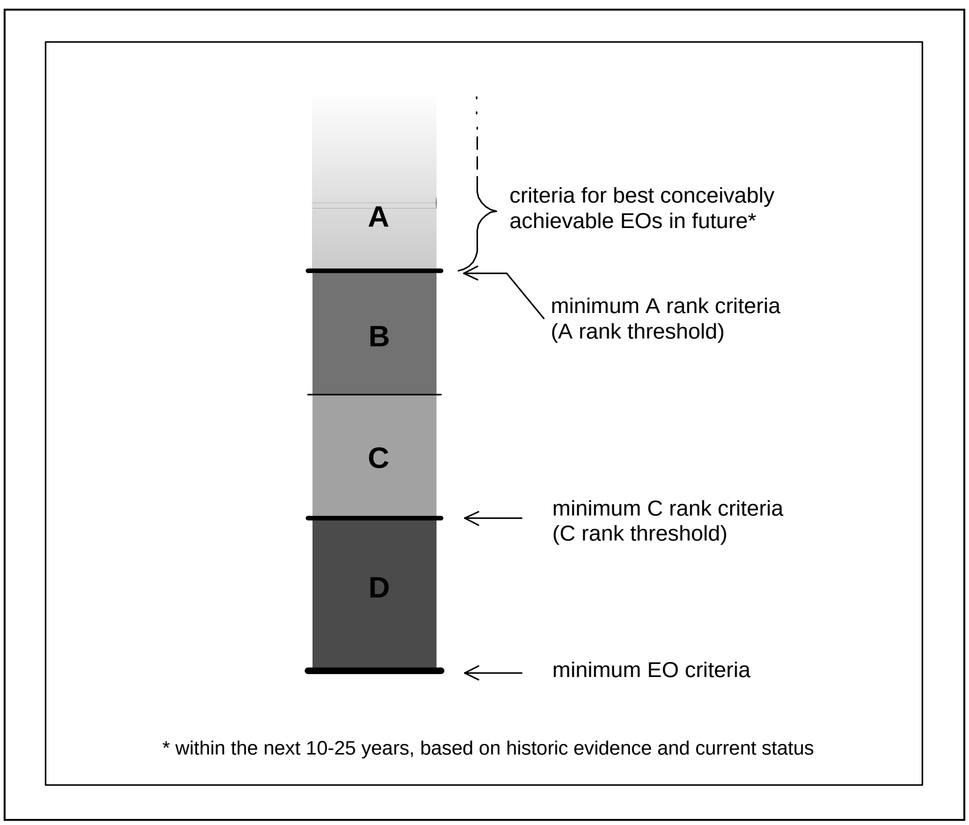


Figure 10. Rank scale for "A", "B", "C", and "D"-ranked EOs.

Especially critical for development of EO rank specifications is the establishment of the threshold between EOs with “fair” and “poor” viability or integrity (the minimum “C” rank criteria). This clarifies whether or not one has a potentially restorable occurrence. Next the A-ranked criteria are established. Typically these are the best EOs that are reasonably and conceivably achievable; generally, these will be the minimum “A” rank criteria unless the best reasonably achievable EOs have only “fair” or “poor” viability or integrity. Finally, assuming the best EOs that are reasonably and conceivably achievable are at or above the “A” rank threshold, one can identify minimum “B” rank criteria that achieve a spread between “A” and “C”-ranked EOs.
An EO rank need not always be directly comparable to historical conditions. For example, bison will not conceivably exist again in their historical condition with herds numbering in the millions; nevertheless a range of viable populations (*e.g.*, herds of differing sizes and conditions) might still be reasonably achievable. In other words, it is still necessary to conceive of a range of viable populations, although the range is truncated when compared to EO rank specifications that would have been written 150 years ago. Similarly, some fire-adapted ecological systems historically supported fire on vast landscape scales that would not be feasibly repeated today. But under controlled conditions, many effects of those landscape scale fires could be reintroduced in smaller areas. These are the types of practical considerations that are documented in EO Rank Specifications. Further details are provided in NatureServe’s (2003) Element Occurrence Data Standards.

Table 7 provide an example where occurrence ranking criteria were established and applied in the Cosumnes River Preserve managed by The Nature Conservancy of California (The Nature Conservancy 2003). In this instance, indicators for a vernal pool system were evaluated. They provided the focus for establishing current status and desired future conditions in this area. These same criteria could be used in other similar examples throughout the range of the ecological system type.

Table 7. Partial EO Rank document for the Northern California Hardpan Vernal Pool *modified from the Consumnes River Preserve Plan of The Nature Conservancy of California.*

Category	Key Ecological Attribute	Indicators	Indicator Ratings **Categorical** Current state: shaded; *Italics* = Desired Rating				Basis for Rating	Viability Objective	Current Status [Date]	Basis for Current Status Rating
			Poor	***Fair***	***Good***	***Very Good***				
Landscape Context	Fire Area-Intensity Regime	Buffer around vernal pool complex that can be fire managed	< 0.25 mile buffer	0.25 - 0.49 mile buffer	0.5 – 0.99 mile buffer	*> 1 mile buffer over >80% of the perimeter of vernal pool properties*	Marty (TNC) 2001	Maintain a buffer of ≥ 1 mile around vernal pool complex on large vernal pool tracts	1 mi buffer intact around Howard and Schneider Ranches (2001)	Analysis of remote sensing data
Landscape Context	Fire Area-Intensity Regime	Fire return interval and area burned	Fire return interval < 1 year or > 10 years for > 10% of the vernal pool grassland.	Fire return interval between 7-10 years for > 10% of the vernal pool grassland.	Fire return interval between 5-7 years for > 50% of the vernal pool grassland.	*Fire return interval between 3-5 years for > 80% of the vernal pool grassland.*	Marty (in prep) 2001; R. Wills pers. comm.; Pollak and Kan 1998; Menke1992	Maintain a prescribed fire return interval of 3-5 years for over 80% of the vernal pool grasslands on the Preserve.	> 10 year fire return interval for > 10% of the Preserve's vernal pool grasslands	Historical fire data
Landscape Context	Connectivity of vernal pool complexes	Distribution of land permanently protected	< 10% connectivity	10-49% connectivity	50-74% connectivity Note: 15-25,000 ac would be protected with this connectivity to be rated Good.	*75% or higher connectivity*	CRP Planning Team 2000; *This Land. Context is linked to the area protected under Size, below.*	Establish 75% connectivity of protected vernal pool habitat by 2005	> 50% connectivity (DE. 2001)	Actual land or easement purchases
Condition	Native species diversity	Native species cover	Relative native species cover (RNSC) in vernal pools < 80%	RNSC in vernal pools 80-84%	RNSC in vernal pools 85-90%	*RNSC in vernal pools >90%*	Monitoring data – Marty (2001)	Maintain relative native species cover >90% in vernal pools	Howard Ranch mean-90%, se=1.7%; Valensin Ranch –Mean=84%, se=3% (2001)	Monitoring data (Marty 2001)
Condition	Native species diversity	native species richness	Richness on pool edge <5 species/quadrat (35 cm x 70 cm)	Richness on pool edge 6-8 species/quadrat	*Richness on pool edge 9-10 species/quadrat*	Richness on pool edge >10 species/quadrat	Monitoring data – Marty (2001)	Maintain average native species richness on the pool edge >10 species/quadrat	Howard Ranch mean-10.4, se=0.32; Valensin Ranch –Mean=9.4, se=0.34 (2001)	Monitoring data (Marty 2001)
Condition	Pollination	overall		?			See regeneration of species but populations are heavily fragmented	Need baseline data to determine quantitative measures for this indicator – hold expert meeting	No information on what or how to measure. Identify experts and hold meeting: 2003.	
Size	Size of vernal pool complexes	Acres of land permanently protected through conservation easement or other	< 10,000 acres protected,	10,000 to 15,000 ac protected	15,000 to 25,000 ac protected Note: This acreage would be protected with 50-74% connectivity to be rated Good.	*30,000 ac protected*	CRP Planning Team 2000; *This Size is linked to the connectivity under Land. Context, above.*	Protect 30,000 ac of vernal pool habitat with 75% in large, contiguous parcels by 2005	17,000 ac protected (Dec. 2001)	Actual land or easement purchases

Applications to Habitat Modeling

Biologists have long used knowledge of an animal's habitat to predict its presence or absence. Numerous approaches to mapping species habitat are well summarized by Scott et al. (2002). Most traditional methods rely only on the location or observation of specimens and include no information on the ecological conditions, such as vegetation and climate variables. Using terrestrial ecological system units as a surrogate to map presence or absence of species habitat has limitations but also provides enhancement over many traditional methods. Because the process would not rely solely on known locality records, unsampled areas can be included in predicted models. Coupling known locations with those predicted from ecological system units, and other ancillary data sets, could lead to more refined maps of species distribution. Given the national scope of this classification, this approach can now be applied consistently across the nation.

Several factors complicate the use of any type of vegetation or habitat map to predict species presence and absence (Scott et al. 2002). For example, birds respond as much or more to vegetation structure than to floristics. NVC alliance units integrate vegetation structure with composition, and have been shown to provide useful predictors of songbird habitat. However, there are also many examples where other environmental factors, such as the presence of steep cliffs or canyons, in association with certain vegetation or water sources, better characterize specific habitat. Species associated with certain hydrologic regimes can be falsely predicted or overestimated unless hydrology and/or riparian habitats are incorporated as linear map features. Habitat for fossorial rodents can also be poorly predicted if vegetation maps do not integrate soil characteristics very well. Terrestrial ecological systems integrate regional climate, local landform, some soil characteristics, as well as local patterns in vegetation and structure into their definition. By mapping ecological system units, many common attributes of wildlife habitat may be better expressed.

Another complication in habitat modeling arises from the variation in specificity of habitat requirements among different species. Some species are generalists in their habitat. Others are restricted to narrow habitat types. In addition, our ability to map certain habitat characteristics can often surpass our knowledge of habitat requirements for many species. As a result, classifications of wildlife habitat vary significantly in the scope and concept of units described. They also vary from state to state, or among different land managing agencies. Ecological system units are more consistently defined in terms of concept and fall into repeatable categories of spatial scale. They may be useful for "crosswalking" among existing habitat classification systems within and across jurisdictions. Appendix 4 includes an example where some 110 ecological system units that fall within California are crosswalked with the 53 wildlife habitat relationship classes of Mayer and Laudenslayer (1988). Given the likelihood that all 110 ecological system units could be mapped across the state of California and into adjacent states, these units should provide significant utility for wildlife assessment.

Avenues for Classification Refinement

As stated previously, ecological classification ideally proceeds through several phases in a continual process of refinement. These phases include 1) literature review and synthesis of current knowledge, 2) formulating initial hypotheses and tentatively describing each type, that support 3) establishing a field sample design, 4) gathering of field data, 5) data analysis and interpretation, 6) description of types, 7) establishing dichotomous keys to classification units, 8) mapping of classification units, and 9) refinement of the classification.

In preceeding sections, we demonstrated how ecological system units can be inventoried and mapped, using existing methodologies and mapped data, at both regional and local scales. These results indicate both the potential utility of ecological system units and a number of directions for their refinement. Mapping ecological systems serves as an immediate test of classification concepts, ensuring that the mapped area is treated comprehensively by the classification, providing for a consistent use of multiple spatial data, and clarifying distinctions between types. Regional mapping provides an initial coverage of system distribution based primarily on the date of remotely sensed imagery. Depending on the ancillary data sets used in map development, these maps may be overlain with other independently derived spatial data, such as elevation, landforms, geology, soils, etc, to further describe the distribution, environmental setting, and landscape patterns that characterize each system type. These maps, if derived using several year-old remotely-sensed imagery, should also function as a practical basis for sample design to gather "training" data for mapping with new imagery.

As noted by Jennings et al. (2003), a vegetation association or community represents a statistical and conceptual synthesis of floristic patterns (Westhoff and van der Maarel 1973, Mueller-Dombois and Ellenberg 1974, Kent and Coker 1992). It is an abstraction, representing a defined range of floristic, structural, and environmental variability. Ecological systems represent a similar kind of abstraction that encompasses the concepts of multiple vegetation associations, and emphasizes the environmental attributes that result in their co-occurrence on the ground. The definition of both associations and ecological systems as individual types is the result of a set of classification decisions based on field sampling, data analysis and interpretation. Suggested approaches to these phases are well summarized by Jennings et al. (2003) for application to vegetation classification.

Two criteria must be met in order for any analysis to be robust. First, the samples must represent a wide range of the compositional, structural, and environmental variation of the proposed type or group of closely related types. Second, there must be a sufficient level of redundancy in the samples to statistically identify mutually exclusive clusters in the data. Standardized approaches for defining ecological system units follows closely from those for vegetation units, but with important caveats. For example, although one should take field sample plots within relatively homogeneous vegetation patches, sampling for ecological system units should consider use of transect-based or other sub-sample plot designs to allow for consistent samples of several component associations *and* document associations in similar environments that make up the ecological system occurrence (see Whittaker 1975).

Measurement of the similarity or dissimilarity among the field samples is central to most classification approaches. A number of quantitative methods for evaluating beta diversity - in terms of turnover in species presence/absence - are commonly applied in vegetation studies (Wilson and Schmida 1984, Magurran 1988), and these could be applied to the data from sub-sample designs. Other quantitative approaches allow for integration of multiple factors, such as relative abundance of vegetation or environmental variables, into more abstract multi-scale information statistics that support analyses better suited to ecosystem classification (Loehle and Wein 1994). Existing data, combined from various sources, are often too heterogeneous to be usable in these quantitative analyses, but such analyses should be considered when designing future sampling.

Conclusions

This report presents work conducted to classify and describe terrestrial ecological systems in the coterminous United States and southern Alaska, and adjacent portions of Mexico and Canada (including coastal British Columbia). A terrestrial ecological system is defined as a group of plant community types (associations) that tend to co-occur within landscapes with similar ecological processes, substrates, and/or environmental gradients. A given terrestrial ecological system will typically manifest itself in a landscape at intermediate geographic scales of 10s to 1,000s of hectares and persist for 50 or more years.

The classification produced here is at a "meso-scale," both spatially and temporally, and the specific spatial and temporal scales are further refined by the biotic and ecological distinctiveness of the system. Our goal was to provide a set of such system types for conservation and resource management applications. Other classifications, which are typically hierarchically arranged, do well at either micro or macro scales. We show how our classification both relies on those efforts and can be linked to them. In fact, the floristic units of the IVC/NVC are an integral part of defining the concepts and spatial limits of the system types. At this time, we focus on a single system level, defined by modular diagnostic classifiers that help to describe the essential ecological and vegetational characteristics of the type. We used an expert-based approach to define a "working set" of system types, and outline further steps for their ongoing development.

This effort resulted in the identification and description of 599 upland and wetland ecological system types within the project area. They represent the full range of natural gradients, with some 381 types (63%) being uplands, 183 types (31%) being wetland, and 35 types (6%) being complexes of uplands and wetlands. Excluding upland/wetland complexes, some 322 types (54%) are predominantly forest, woodland, and/or shrubland, and some 166 types (28%) are predominantly herbaceous, savanna, or shrub steppe. Seventy-four types (12%) are sparsely vegetated or "barren."

Terrestrial ecological system units provide practical, systematically defined groupings of plant community types that can enhance the mapping of terrestrial communities and ecosystems at multiple scales of spatial and thematic resolution. We provide a number of applications of ecological system units to conservation assessment, ecological inventory, mapping, land management, ecological monitoring, and species habitat modeling. The classification, referred to as the U.S. Terrestrial Ecological Systems Classification, is the U.S. component of an International Terrestrial Ecological Systems Classification. NatureServe and partners will facilitate continued development and refinement of this classification.

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Appendices

Appendix 1. Existing Classification Systems

The Ecological Systems Classification draws heavily on concepts and units from previously established classification systems, some of which are "multi-factor" classifications (vegetation, landform, soil, etc.) while others take a "single factor" approach (e.g. vegetation only). A brief review of selected classification approaches provides additional background useful for comparison and contrast with the multi-factor approach taken here to define ecological system units.

State Natural Heritage Program Community Classifications: The "natural community type" concept has been widely used develop state-level classifications, defining units by a combination of criteria, including vegetation physiognomy, current species composition, soil moisture, substrate, soil chemistry, or topographic position, depending on the local situation (e.g. Reschke 1990, Schafale and Weakley 1990). This approach has been used with great success for conservation and inventory at the local and state level, but there have been no consistent rules for defining "natural community" concepts for applicability at broader scales.

Ecological Site Classification: There are a number of classification approaches that combine abiotic and biotic criteria at various scales for classifying landscape ecosystems, ecological land units, or site types (e.g., Barnes 1984, McNab and Avers 1994, Avers et al. 1994). Beginning as early as the Life Zone classifications of Merriam (1898), site classifications use physiographic or environmental characteristics along with vegetation. Ecological land classification approaches integrate climate, physiography, landform, soil, and vegetation to define ecosystem or ecological land units, typically within a spatially nested hierarchy (e.g. Lapin and Barnes 1995, Bailey 1996). The products of these efforts often include type descriptions along with maps. While data intensive, these classifications have been developed throughout many forested portions of the United States and have often been used to guide forest management.

In practice, landscape-based approaches have been extremely useful for defining regional landscape ecosystems, or ecoregions, that serve as a useful spatial framework for conservation assessment (Bailey 1998, Barnes et al.1998). They also tend to be quite valuable at very local scales (<10s of hectares) to describe site potential for intensive management and monitoring (Cleland et al. 1998). However, only the finest scale ecological land types could practically be said to recur across a given regional landscapes. Intermediate scale landscape units (e.g. "land type associations") tend to include considerable ecological heterogeneity. One would be hard-pressed to describe mid-scale landscape units as truly "recurring" landscape features. They are often best considered unique units with varying levels of similarity with other unique units. This aspect limits their utility for some conservation applications.

Habitat Type Classification: The habitat-type approach, applied extensively by the U.S. Forest Service (Wellner 1989), relies primarily on species occurrence criteria and concepts of potential natural vegetation to define site types or habitat types. Potential natural vegetation is often defined as "the vegetation structure that would become established if all successional sequences were completed without interference by man under the present climatic and edaphic conditions (including those created by man)" (Tüxen 1956, in Mueller-Dombois and Ellenberg 1974). Late successional dominants are used to organize types along an elevational gradient from grassland to alpine tundra. Habitat type classifications typically include dichotomous keys to each unit. Because these classifications integrate environmental factors such as climate and soil characteristics, they may be broadly applied for recurring map units across regional landscapes. However, they share a weakness with ecological site classifications in that they seldom can fully integrate factors of landscape juxtaposition that effect prevailing disturbance regimes and the existing vegetation one would encounter on the ground. Analysis of historical land cover data has indicated the significance of this factor in several regions of the United States (e.g. Comer et al. 1995).

Natural Resource Conservation Service Ecological Sites (*http://plants.usda.gov/esis*). In this approach, soil is the basis for determining, correlating, and differentiating one ecological site from another. Soils with like properties that produce and support a characteristic native plant community, and that respond similarly to management, are grouped into the same ecological site. Criteria used differentiate one ecological site from another include a) significant differences in the species or species groups that are in the characteristic plant community, b) significant differences in the relative proportion of species or species groups in the characteristic plant community, c) soil factors that determine plant production and composition, the hydrology of the site, and the functioning of the ecological processes of the water cycle, mineral cycles, and energy flow, and d) differences in the kind, proportion, and production of the overstory and understory plants due to differences in soil, topography, climate, and environment factors, or the response of vegetation to management.

In practice, ecological sites may define units at or near the scale of plant associations of the National Vegetation Classification (see below), or small groups of associations.

The National Wetland Classification System (Cowardin et al. 1979): This classification forms the basis for the USDI National Wetland Inventory Classification and Mapping Program. In this system, the hierarchical levels are defined by water body types (marine, riverine, palustrine), substrate materials, flooding regimes, and vegetation life forms. The lowest unit is the dominance type, named for the dominant plant and animal forms, and is developed by the user, so it varies with each application. This system can be mapped, but some features, such as flooding regimes are very dynamic and multi-temporal observation is often required.

HGM, or Hydrogeomorphic Approach: The HGM Approach is a multi-agency effort involving the U.S. Army Corps of Engineers, the Environmental Protection Agency, the Federal Highway Administration, the Natural Resources Conservation Service, and the U.S. Fish and Wildlife Service. This approach is intended to support methods for assessing the physicial, chemical, and biological functions of wetlands (Brinson 1993). It is based on wetland hydrogomorphic properties of geomorphic setting, water source, and hydrodynamics. A suite of indicators are used to describe each of these properties then develop "profiles" that describe the functions the wetland is likely to perform. While of great utility for its intended purpose, the HGM approach is not designed to be sensitive to species composition of vegetation.

North American Biotic Communities are described using a biogeographic approach (Brown et al. 1998). This classification is formulated on the limiting effects of moisture and temperature minima on the structure and composition of vegetation as well as the specific plant and animal adaptations to regional environments. It draws on a long history of defining regional biomes, taking into account regional patterns in both plant and animal distributions to define communities at varying hierarchical scales (e.g. Udvardy 1975; Brown, Lowe, and Pase 1980). A six-level hierarchy is used to describe these types (Table 1.1). This results in some 150 Biotic Community units across the coterminous United States. The potential distribution of some 36 biotic community types were also mapped (Reichenbacher et al. 1999).

This approach provides many useful insights for biogeographic regionalization and the application of biogeographic criteria to make practical inferences for the likely biotic composition of communities in a given regional landscape. However, not unlike the National Vegetation Classification (see below) there is a considerable break in the number of classification units between, for example, the Biotic Community scale and the Series scale, the latter of which likely includes over 1,000 units in the coterminous United States, if fully developed.

Table 1.1. Hierarchical Structure for Biotic Community Classification System

Hydrologic Regime (Upland vs.Wetland)
Formation Type (Swamp and Riparian Forest, Swamp and Riparian Scrub, Marshland, Strand, Submergent)
Climate Zone (Arctic-Boreal, Cold Temperate, Warm Temperate, Tropical-Subtropical)
Biogeographic Province (Northeastern, Plains, Rocky Mountain, Great Basin, Sierra-Cascade, Oregonian)
Biotic Community (e.g. Great Basin Interior Marshland)
Series (e.g. Bulrush Series)
Association (e.g. *Scirpus paludosus* Association)

European EUNIS Habitats and Phytosociological Classification: This is a standardized habitat classification describing some 1,200 natural ecological units for the European continent, integrating environmental factors with predominant vegetation (Davies and Moss 1999). These habitats are arranged in a simple hierarchical structure with Table 1 including the upper-most set of units in the hierarchy.

Table 1.2. EUNIS Habitat Classification

EUNIS Habitat Classification, Level 1
Marine habitats
Coastal habitats
Inland surface water habitats
Mire, bog and fen habitats
Grassland and tall forb habitats
Heathland, scrub and tundra habitats
Woodland and forest habitats and other wooded land
Inland unvegetated or sparsely vegetated habitats

The long tradition of phytosociology throughout Europe has been recently integrated with the EUNIS habitat classification, linking 928 Alliance units to each EUNIS habitat (Rodwell et al. 2002).

U.S. National Vegetation Classification. The NVC was established as the standard classification framework for vegetation by federal agencies in the United States (FGDC 1997). The following basic tenets underlie the terrestrial portion of the NVC:

1. The NVC is based primarily on vegetation, rather than soils, landforms or other non-biologic features.

This was decided upon mainly because plants are easily measured biological expressions of environmental conditions and are directly relevant to biological diversity. Vegetation is complex and continuously variable, with species forming only loosely repeating assemblages in ecologically similar habitats. The NVC does not solve the problems inherent in any effort to categorize the continuum of vegetation pattern, but it presents a practical set of methods to bring consistency to the description of vegetation.

2. The NVC applies to all terrestrial vegetation. In addition to upland vegetation, “terrestrial vegetation” is defined to include all wetland vegetation with rooted vascular plants. It also includes communities characterized by sparse to nearly absent vegetation cover, such as those found on boulder fields or talus.

3. The NVC focuses on existing vegetation rather than potential natural or climax vegetation.

The vegetation types described in the classification range from the ephemeral to the stable and persistent. Recognizing and accommodating this variation is fundamental to protecting biodiversity. The manner in which a community occurs is, in part, an intrinsic property of the vegetation itself. A classification that is not restricted to static vegetation types ensures that the units are useful both for inventory/site description, and as the basis for building dynamic ecological models.

The current scope of the NVC includes:

1. While the NVC framework can be used to classify all vegetation, emphasis has been given to vegetation types that are natural or near-natural, i.e., those that appear to be unmodified or only marginally impacted by human activities. Where anthropogenic impacts are apparent, the resulting physiognomic and floristic patterns have a clear, naturally-maintained analog.

2. Classification development at the finest levels of the system has so far focused on the contiguous United States and Hawaii. Some classification at finer levels has also been done for southeastern Alaska, parts of Canada, the Caribbean, and a few areas in northern Mexico.

NVC HIERARCHY

The top division of the classification hierarchy separates vegetated communities (Terrestrial System) from those of unvegetated deepwater habitats (Aquatic System) and unvegetated subterranean habitats (Subterranean System). The Terrestrial System is broadly defined to include areas with rooted submerged vegetation of lakes, ponds, rivers, and marine shorelines, as well as the vegetation of uplands.

The hierarchy for the vegetated communities has seven levels: the five highest (coarsest) levels are physiognomic and the two lowest (finest) levels are floristic. The levels of the terrestrial classification system are listed and described below.

VEGETATION CLASSIFICATION SYSTEM

	FORMATION CLASS
	FORMATION SUBCLASS
	FORMATION GROUP
	FORMATION SUBGROUP
physiognomic levels	FORMATION
floristic levels	ALLIANCE
	ASSOCIATION

PHYSIOGNOMIC LEVELS

The physiognomic portion of the NVCS hierarchy is a modification of the UNESCO world physiognomic classification of vegetation (1973) and incorporates some of the revisions made by Driscoll et al. (1984) for the United States. Details of the hierarchy are described in Grossman et al. (1998). The lowest physiognomic level is the formation.

Formation

The formation represents a grouping of community types that share a definite physiognomy or structure and broadly defined environmental factors, such as elevation and hydrologic regime. Structural factors such as crown shape and lifeform of the dominant lower stratum are used in addition to the physiognomic characters already specified at the higher levels. The hydrologic regime modifiers were adapted from Cowardin et al. (1979). Examples include: Rounded-crowned temperate or subpolar needle-leaved evergreen forest, Seasonally flooded cold-deciduous forest, Semipermanently flooded cold-deciduous shrubland, Tall sod temperate grassland, Cliffs with sparse vascular vegetation.

FLORISTIC LEVELS

Alliance

The alliance is a physiognomically uniform group of plant associations (see association below) sharing one or more dominant or diagnostic species, which as a rule are found in the uppermost strata of the vegetation (Grossman et al. 1998). Dominant species are often emphasized in the absence of detailed floristic information (such as quantitative plot data), whereas diagnostic species (including characteristic species, dominant differential, and other species groupings based on constancy) are used where detailed floristic data are available (Moravec 1993).

For forested communities, the alliance is roughly equivalent to the "cover type" of the Society of American Foresters (Eyre 1980), developed for use primarily by foresters to describe the forest types of North America. The alliance may be finer in detail than a cover type when the dominant tree species extend over large geographic areas and varied environmental conditions (e.g. the *Pinus ponderosa* Forest Alliance, *Pinus ponderosa* Woodland Alliance, and *Pinus ponderosa* Temporarily Flooded Woodland Alliance are all within the *Pinus ponderosa* Cover Type of the SAF). Alliances, of course, have also been developed for non-forested vegetation.

The alliance is similar in concept to the "*series*," as developed for the Habitat Type System to group habitat types that share the same dominant species under "climax" conditions (Daubenmire 1952, Pfister and Arno 1980). Alliances, however, are described by the dominant or diagnostic species for *all* existing vegetation types, whereas series are generally restricted to potential "climax" types and are described by the primary dominant species.

Association

The association is the lowest level, as well as the basic unit for vegetation classification, in the NVCS. The association is defined as "a plant community of definite floristic composition, uniform habitat conditions, and uniform physiognomy" (see Flahault and Schroter 1910 in Moravec 1993). This basic concept has been used by most of the schools of floristic classification (Whittaker 1962, Braun-Blanquet 1965, Westhoff and van der Maarel 1973, Moravec 1993).

The plant association is differentiated from the alliance level by additional plant species, found in any stratum, which indicate finer scale environmental patterns and disturbance regimes. This level is derived from analyzing complete floristic composition of the vegetation unit when plot data are available. In the absence of a complete data set, approximation of this level is reached by using available information on the dominant species or environmental modifiers, and their hypothesized indicator species.

Table 1.3. Three Examples from the National Vegetation Classification Hierarchy

CLASS	FOREST	WOODLAND	SHRUBLAND
SUBCLASS	Deciduous Forest	Evergreen Woodland	Deciduous Shrubland
GROUP	Cold-deciduous Forest	Temperate or Subpolar Needle-leaved Evergreen Woodland	Temperate Broad-leaved Evergreen Shrubland
SUBGROUP	Natural/Semi-natural	Natural/Semi-natural	Natural/Semi-natural
FORMATION	Lowland or Submontane Cold-deciduous Forest	Saturated Temperate or Subpolar Needle-leaved Evergreen Woodland	Sclerophyllous Temperate Broad-leaved Evergreen Shrubland
ALLIANCE	*Quercus stellata - Quercus marilandica* Forest Alliance	*Pinus palustris* Saturated Woodland Alliance	*Quercus havardii* Shrubland Alliance
ASSOCIATION	*Quercus stellata - Quercus marilandica - Carya (glabra, texana) / Vaccinium arboreum* Forest	*Pinus palustris / Leiophyllum buxifolium / Aristida stricta* Woodland	*Quercus havardii - (Penstemon ambiguus, Croton dioicus) / Sporobolus giganteus* Shrubland

Appendix 2. Element Occurrence Specifications

Elements, the basic components of biodiversity tracked by NatureServe and its natural heritage program members, include species, communities, and ecosystems. Element Occurrence Specifications provide the methodology for deciding when two or more mapped polygons of an element represent a single occurrence) (see Stein and Davis 2000). Methods previously developed for community occurrences apply with limited modification to ecological systems (NatureServe 2003).

General Guidelines

Minimum criteria.

For communities and systems, minimum criteria for EOs are implicit in the classification of the Element. A brief description of the Element (*e.g.,* composition, structure, ecological processes, component associations) that includes information on characteristics that distinguish it from similar communities or systems should be provided in a global Element summary field. Any area that is large enough to be classified as a particular community or system Element has, in essence, met the minimum criteria for an occurrence of that type. Practically, however, minimum sizes may be helpful and should be provided in the EO specifications.

Note that the minimum EO requirement is not based on the C/D threshold. Otherwise, all D-ranked EOs are, by definition, not EOs. Thus a System label could be applied to a small 10 ha stand of Shortleaf Pine-Hardwood Matrix System in an agricultural landscape. It may not be viable, and it may be that Network ecologists would not document the EO (unless it was a very rare community or system), but it could still be an EO. It is important to distinguish issues of EO-Tracking versus minimum EO specs. The minimum size is the smallest size of a component "core association" or cluster of associations that is recognizable (classifiable) as a System Element.

Recommended minimum sizes for the different community pattern types are:

- 2 hectares for matrix;
- 0.4 hectare for large patch;
- 0.05 hectare for small patch; and
- 30 meters in length for linear.

Recommended minimum sizes for the system types will meet or exceed those of the component community types. They are:

- 10 ha for matrix,
- 10 ha for upland large patch;
- 1 ha for wetland large patch;
- 0.5 ha for small patch;
- 100 m for all linear types.

Stands/areas below the recommended minimum size become difficult to judge in terms of community or system type characteristics, and, if isolated, become heavily influenced by edge effects. For conservation purposes, generally only larger sized occurrences of each community type are tracked and the threshold for minimum size is seldom approached.

Separating EOs:

Principal EOs are typically separated from other principal EOs, either by barriers or breaks, or by specified distances across intervening areas. For communities or systems, separation distances will be measured across intervening areas of different natural or semi-natural communities, or cultural vegetation based on their effect on ecological processes or species interactions.

Barriers

Known barriers for Elements, either naturally occurring or manmade, should be described in the EO specifications. For community or system EOs, barriers may be obstacles that limit the expansion or alter the function of these types. These barriers either separate populations of most of the component species within the community or system, thus obstructing or severely limiting gene flow and ecological interactions or they obstruct or limit ecological processes that these species depend on. Barriers may be common for many aquatic and wetland communities or systems, but are typically less common for many upland terrestrial communities or systems.

Separation Distances

In addition to barriers that totally, or almost completely, prevent ecological processes and species interactions, there may be habitats between two stands of an element that partially restrict species interactions or ecological processes. Unlike barriers, their effect depends on the kind and extent of this intervening habitat and its effect on the stands. This leads to the issue of separation distance. The intent of assigning values for separation distances between two stands is to achieve consistency in the manner in which EOs are defined and mapped. Thus, smaller separation distances are used when the intervening habitat is highly restrictive to the ecological processes or species interactions the element depends, and greater distances are used when these habitats are less prohibitive to ecological processes or species interactions.

We use two broad categories of intervening habitats to define separation distances, namely – natural/semi-natural vegetation or cultural vegetation. Generally speaking, intervening natural and semi-natural vegetation will have less of an ecological effect between two stands of an EO than intervening cultural vegetation. Thus rather simplistically, we suggest that different separation distances be specified for these two kinds of situations. Typically, a shorter separation distance is specified when the intervening habitat is cultural vegetation than when it is natural/semi-natural. Minimum values for separation distances have been recommended to ensure that EOs are not separated by unreasonably small distances, which would lead to the identification of unnecessarily splintered stands as potential targets for conservation planning or action. For communities or systems, the minimum separation distance for intervening areas of different natural or semi-natural communities is set at 1 km or greater, and for intervening areas of cultural vegetation, the distance is set at 0.5 km or greater.[4] Table 2.1 summarizes the recommended minimum separation distances for community and system EOs. These separation distances may, of course, be much larger. For communities or systems found primarily in mountainous regions, where habitat tends to be less fragmented, separation distances may be 5 km or more.

It is possible that these separation distances could be further refined by considering the kind of natural/semi-natural or cultural vegetation present. Intervening natural and semi-natural areas with similar kinds of habitat characteristics to the stands of a community or system under consideration will have less of an effect on community or system processes than those with very different kinds of characteristics. For example, bog stands separated by intervening areas of upland jack pine on bedrock could be more readily treated as distinct EOs than bogs separated by areas of black spruce swamp. However, at this time, no specific guidelines are suggested for these situations, but if used, they should be documented.

[4] Minimum distances for systems are no less than, and may exceed, that of communities. Further review of their recommendations are needed.

Table 2.1 - Recommended Minimum Separation Distances for Communities and Ecological Systems

Type of Separation	Minimum Separation Distance
Barrier	qualitatively defined
cultural vegetation	≥ 0.5 km
different natural or semi-natural communities or systems	≥ 1 km

Example

ELEMENT
North-Central Interior Dry-Mesic Oak Forest and Woodland System (CES202.046)

SPECS GROUP
None

MINIMUM CRITERIA
This system is found throughout the glaciated regions of the Midwest, typically in gently rolling landscapes. It can occur on uplands within the prairie matrix and near floodplains, or on rolling glacial moraines and among kettle-kame topography. Soils are typically well-drained Mollisols or Alfisols that range from loamy to sandy loam in texture. Historically, this type was quite extensive in MI, IN, IL, MO, IA, WI, and MN. Well over 700,000 hectares likely occurred in southern Michigan alone *circa* 1800. It is distinct from other forested systems within the region by a dry-mesic edaphic condition that is transitional between dry oak forests and woodlands and mesic hardwood forests, such as maple-basswood forests. Forest cover can range from dense to moderately open canopy and there is commonly a dense shrub layer. Fire-resistant oak species, in particular *Quercus macrocarpa, Q. rubra,* and/or *Q. alba* dominate the overstory. *Carya* spp., including *C. ovata, C. cordiformis,* and *C. tomentosa* are diagnostic in portions of the range of this system. Depending on range of distribution, and overstory canopy density, the understory may include species such as *Corylus americana, Amelanchier* spp., *Maianthemum stellatum, Caulophyllum thalictroides, Laportea canadensis, Trillium grandiflorum, Aralia nudicaulis,* and *Urtica dioica*. Occasionally, prairie grasses such as *Andropogon gerardii* and *Panicum virgatum* may be present. Fire constitutes the main natural process for this type and likely maintained a more open canopy structure to support oak regeneration. Historic fire frequency was likely highest in the prairie-forest border areas. Fire suppression may account for the more closed oak forest examples of this system with the more mesic understory. It likely has allowed for other associates such as *Acer saccharum, Celtis occidentalis, Liriodendron tulipifera, Ostrya virginiana,* and *Juglans nigra* to become more prevalent, especially in upland areas along floodplains. Extensive conversion for agriculture has fragmented these systems. Continued fire suppression has also resulted in succession to mesic hardwoods, such that in many locations, no oak species are regenerating. Remaining large areas of this system are likely under considerable pressure due to conversion to agriculture, pastureland, and urban development.

Minimum Size: 10 ha

EO Separation

SEPARATION BARRIERS
Barriers that would separate patches of this community include a four-lane highway, urban development, and an open body of water or large river. The open bodies of water or river may act as a fire-break.

SEPARATION DISTANCE – NATURAL/SEMI-NATURAL VEGETATION
4 km
SEPARATION DISTANCE – CULTURAL VEGETATION
0.5 km

ALTERNATE SEPARATION PROCEDURE

SEPARATION JUSTIFICATION
The separation factors for natural/semi-natural vegetation reflect the relatively ease with which species and processes move between systems in the relatively flat glaciated landscape. In addition, seed dispersal of *Quercus* and *Carya* spp., which are dependent on squirrels and jays. These dispersers can move considerable distances between patches in intact or fragmented landscapes, from several hundred meters to 4 or 5 km (Harrison and Werner 1984, Crow 1988, Johnson and Webb 1989).

Separation distance for cultural vegetation is set at minimum default value.

FEATURE LABELS

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GSPECS NOTES

Distinctions *within* Element Occurrences.

Although the EO conceptually represents the entire occupied area, there may be smaller geographically distinct areas *within* the principal EO for which information could be useful for conservation planning, biological monitoring, or biological management at local levels. These geographically nested components are referred to as sub-EOs, and the main EO is referred to as the Principal EO. Sub-EOs must be contained within a principal EO of the **same** Element. Note that sub-EOs should not be created simply to represent different parts of a principal EO comprised of noncontiguous patches.

Sub-EOs may be defined as

a) areas of differing composition, or higher density, quality, or conservation concern (*e.g.,* different age stands or successional phases, old growth patches);

b) discrete areas for which it is desirable to maintain information for each area in separate records (*e.g.*, to facilitate recording of monitoring data); or

c) other areas marked by non-biological divisions assigned for convenience in mapping, monitoring, or management (*e.g.,* geographic, political, and land survey map units). The creation of sub-EOs defined by these divisions should generally be avoided because they are not biologically significant.5

Sub-EOs can be used to facilitate information management in cases where a principal EO is particularly large, complex, or crosses jurisdictional boundaries. Such principal EOs may present challenges, including

[5] Some geographic units, such as watersheds, may sometimes reflect biological divisions, particularly for many freshwater Elements.

incomplete knowledge of the full extent of the EO, loss of detail about specific sub-populations or community patches, and difficulty in supporting information needs related to inventory, monitoring, management, conservation planning, and environmental review. However, sub-EOs should not replace the use of a principal EO to represent the full extent of the occurrence.

Community-level EOs should not be treated as sub-EOs of System EOs, as they are two different classification systems, and each level can exist independent of the other (unlike the EO – sub-EO relationship). Doing so would also complicate the ability to track sub-EO features listed above at either level. However, where a community-level EO is a spatial component of a System EO, it is desirable to attribute the community EO with the System EO code in order to display their relationships.

Appendix 3. NatureServe Global Conservation Status Definitions

The Global (G) Conservation Status (Rank) of a species or ecological community is based on the *range-wide* status of that species or community. The rank is regularly reviewed and updated by experts, and takes into account such factors as number and quality/condition of occurrences, population size, range of distribution, population trends, protection status, and fragility. The definitions of these ranks, which are not to be interpreted as legal designations, are as follows:

GX **Presumed Extinct**: Not located despite intensive searches and virtually no likelihood of rediscovery
GH **Possibly Extinct**: Missing; known only from historical occurrences but still some hope of rediscovery
G1 **Critically Imperiled**: At high risk of extinction due to extreme rarity (often 5 or fewer occurrences), very steep declines, or other factors.
G2 **Imperiled**: At high risk of extinction due to very restricted range, very few populations (often 20 or fewer), steep declines, or other factors.
G3 **Vulnerable**: At moderate risk of extinction due to a restricted range, relatively few populations (often 80 or fewer), recent and widespread declines, or other factors.
G4 **Apparently Secure**: Uncommon but not rare; some cause for long-term concern due to declines or other factors.
G5 **Secure**: Common; widespread and abundant.

G(#)**T**(#): Trinomial (T) rank applies to subspecies or varieties; these taxa are T-ranked using the same definitions as the G-ranks above.

Variant Global Ranks

G#G# **Range Rank**: A numeric range rank (e.g., G2G3) is used to indicate uncertainty about the exact status of a species or community. Ranges cannot skip more than one rank (e.g., GU should be used rather than G1G4).
GU **Unrankable**: Currently unrankable due to lack of information or due to substantially conflicting information about status or trends. NOTE: Whenever possible, the most likely rank is assigned and the question mark qualifier is added (e.g., G2?) to express uncertainty, or a range rank (e.g., G2G3) is used to delineate the limits (range) of uncertainty.
GNR **Not ranked**: Global rank not assessed.

Rank Qualifiers

? **Inexact Numeric Rank**: Denotes inexact numeric rank.
Q **Questionable taxonomy that may reduce conservation priority**: Distinctiveness of this entity as a taxon at the current level is questionable; resolution of this uncertainty may result in change from a species to a subspecies or hybrid, or inclusion of this taxon in another taxon, with the resulting taxon having a lower-priority (numerically higher) conservation status rank.

Appendix 4. Terrestrial Ecological Systems and Wildlife Habitats in California

System Code	Terrestrial Ecological System Name	California WHR Classes
	Mainly Wetland	
CES302.759	Sonoran Fan Palm Oasis	Palm oasis
CES304.780	Inter-Mountain Basins Greasewood Flat	Desert riparian
CES206.944	Mediterranean California Foothill and Lower Montane Riparian Woodland	Montane riparian/Valley foothill riparian
CES206.945	Mediterranean California Serpentine Foothill and Lower Montane Riparian Woodland and Seeps	Montane riparian
CES206.946	California Central Valley Riparian Woodland and Shrubland	Valley foothill riparian/Valley oak woodland
CES300.729	North American Arid West Emergent Marsh	Freshwater emergent wetland
CES302.747	North American Warm Desert Cienega	Freshwater emergent wetland
CES302.748	North American Warm Desert Lower Montane Riparian Woodland and Shrubland	Montane riparian
CES302.752	North American Warm Desert Riparian Mesquite Bosque	Desert riparian
CES302.753	North American Warm Desert Riparian Woodland and Shrubland	Desert riparian
CES302.755	North American Warm Desert Wash	Desert dry wash
CES304.768	Columbia Basin Foothill Riparian Woodland and Shrubland	Montane riparian
CES200.876	Temperate Pacific Freshwater Aquatic Bed	Freshwater emergent wetland
CES200.877	Temperate Pacific Freshwater Emergent Marsh	Freshwater emergent wetland
CES204.880	North Pacific Maritime Tidal Salt Marsh	Saline emergent marsh
CES206.947	Mediterranean California Alkali Marsh	Freshwater emergent wetland
CES206.948	Northern California Claypan Vernal Pool	Annual grassland
CES206.949	Northern California Volcanic Vernal Pool	Annual grassland
CES206.950	South Coastal California Vernal Pools	Annual grassland
CES206.951	Mediterranean California Coastal Interdunal Wetland	Freshwater emergent wetland
CES206.952	Mediterranean California Subalpine-Montane Fen	Freshwater emergent wetland
CES206.953	Mediterranean California Serpentine Fen	Freshwater emergent wetland
CES206.954	California Central Valley Alkali Sink	Freshwater emergent wetland
CES204.996	Modoc Basalt Flow Vernal Pools	Annual grassland
CES200.997	Temperate Pacific Brackish Marsh	Estuarine
CES200.998	Temperate Pacific Montane Wet Meadow	Wet Meadow
CES206.999	Mediterranean California Eel Grass Beds	Marine
CES206.002	Mediterranean California Coastal Salt Marsh	Saline emergent marsh
CES304.045	Great Basin Foothill and Lower Montane Riparian Woodland and Shrubland	Montane riparian
CES302.751	North American Warm Desert Playa	Alkali desert scrub
CES304.781	Inter-Mountain Basins Greasewood Wash	Desert dry wash
CES304.786	Inter-Mountain Basins Playa	Alkali desert scrub
CES200.878	Temperate Pacific Freshwater Mudflat	
	Mainly Upland	
CES302.741	Mogollon Chaparral	Mixed chaparral
CES302.742	Mojave Mid-Elevation Mixed Desert Scrub	Joshua tree
CES302.749	Sonora-Mojave Desert Mixed Salt Desert Scrub	Alkali desert scrub
CES302.756	Sonora-Mojave Creosotebush-White Bursage Desert Scrub	Desert scrub

System Code	Terrestrial Ecological System Name	California WHR Classes
CES302.757	Sonora-Mojave-Baja Semi-Desert Chaparral	Mixed chaparral
CES302.760	Sonoran Granite Outcrop Desert Scrub	Desert scrub
CES302.761	Sonoran Paloverde-Mixed Cacti Desert Scrub	Desert succulent scrub
CES304.769	Columbia Plateau Western Juniper Savanna	Juniper
CES304.772	Inter-Mountain Basins Mountain Mahogany Woodland and Shrubland	
CES304.773	Great Basin Pinyon-Juniper Woodland	Pinyon-juniper
CES304.774	Great Basin Xeric Mixed Sagebrush Shrubland	Sagebrush
CES304.777	Inter-Mountain Basins Big Sagebrush Shrubland	Sagebrush
CES304.778	Inter-Mountain Basins Big Sagebrush Steppe	Sagebrush
CES304.782	Inter-Mountain Basins Juniper Savanna	Juniper
CES304.784	Inter-Mountain Basins Mixed Salt Desert Scrub	Alkali desert scrub
CES304.785	Inter-Mountain Basins Montane Sagebrush Steppe	Sagebrush
CES304.787	Inter-Mountain Basins Semi-Desert Grassland	Perennial grassland
CES304.788	Inter-Mountain Basins Semi-Desert Shrub Steppe	Perennial grassland
CES304.789	Inter-Mountain Basins Shale Badland	Alkali desert scrub
CES304.790	Inter-Mountain Basins Subalpine Limber-Bristlecone Pine Woodland	Subalpine conifer
CES306.813	Rocky Mountain Aspen Forest and Woodland	Aspen
CES204.852	North Pacific Oak Woodland	Montane hardwood
CES206.900	Mediterranean California Alpine Fell-Field	Low sagebrush
CES206.909	Mediterranean California Mixed Oak Woodland	Montane hardwood
CES206.910	Mediterranean California Subalpine Woodland	Subalpine conifer
CES206.911	Northern Pacific Mesic Subalpine Woodland	Subalpine conifer
CES206.912	Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland	Lodgepole pine
CES206.913	Mediterranean California Red Fir Forest and Woodland	Red fir
CES206.914	Klamath-Siskyou Upper Montane Serpentine Mixed Conifer Woodland	Klamath mixed conifer
CES206.915	Mediterranean California Mesic Mixed Conifer Forest and Woodland	Sierran mixed conifer forest/White fir/Douglas fir
CES206.916	Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland	Sierran mixed conifer forest/White fir/Douglas fir
CES206.917	Klamath-Siskyou Lower Montane Serpentine Mixed Conifer Woodland	Klamath mixed conifer
CES206.918	Mediterranean California Ponderosa-Jeffrey Pine Forest and Woodland	Ponderosa pine/Jeffrey pine/Eastside pine
CES206.919	Northern California Mixed Evergreen Forest	Montane hardwood/Douglas fir
CES206.920	Central and Southern California Mixed Evergreen Woodland	Montane hardwood
CES206.921	Coastal Redwood-Mixed Conifer Forest and Woodland	Redwood/Douglas fir
CES206.922	Coastal Closed-Cone Conifer Forest and Woodland	Closed-cone pine-cypress
CES206.923	Mediterranean California Mixed Oak-Evergreen Woodland	Montane hardwood - conifer
CES206.924	Sierra Nevada Alpine Dwarf Shrubland	Alpine dwarf shrub/Low sagebrush
CES206.925	California Montane Woodland and Chaparral	Montane chaparral
CES206.926	California Mesic Chaparral	Mixed chaparral
CES206.927	California Xeric Serpentine Chaparral	Mixed chaparral
CES206.928	Mesic Serpentine Woodland and Chaparral	Mixed chaparral
CES206.929	California Maritime Chaparral	Mixed chaparral
CES206.930	Southern California Dry-Mesic Chaparral	Chamise-red shank

System Code	Terrestrial Ecological System Name	California WHR Classes
CES206.931	Northern and Central California Dry-Mesic Chaparral	Mixed chaparral
CES206.932	Northern California Coastal Scrub	Coastal scrub
CES206.933	Southern California Coastal Scrub	Coastal scrub
CES206.934	Baja Semi-Desert Coastal Succulent Scrub	Desert succulent scrub
CES206.935	California Central Valley Mixed Oak Savanna	Blue oak woodland/Valley oak woodland
CES206.936	California Lower Montane Pine-Oak Woodland and Savanna	Blue oak-Digger pine
CES206.937	California Coastal Live Oak Woodland and Savanna	Coastal oak woodland
CES206.938	Southern California Oak Woodland and Savanna	Coastal oak woodland
CES206.939	Mediterranean California Alpine Dry Tundra	Perennial grassland
CES206.940	Mediterranean California Subalpine Meadow	Perennial grassland
CES206.941	California Northern Coastal Grassland	Perennial grassland
CES206.942	California Central Valley and Southern Coastal Grassland	Perennial grassland
CES206.943	California Mesic Serpentine Grassland	Perennial grassland
CES204.100	North Pacific Montane Grassland	Perennial grassland
CES304.001	Great Basin Semi-Desert Chaparral	Mixed chaparral
CES304.042	Great Basin Altered Andesite Pine Woodland	Ponderosa pine/Jeffery pine
	Mainly Sparsely Vegetated	
CES302.744	North American Warm Desert Active and Stabilized Dunes	Desert scrub
CES302.745	North American Warm Desert Bedrock Cliff and Outcrop	
CES302.750	North American Warm Desert Pavement	
CES302.754	North American Warm Desert Volcanic Rockland	
CES304.779	Inter-Mountain Basins Cliff and Canyon	
CES206.899	Mediterranean California Alpine Bedrock and Scree	
CES206.901	Sierra Nevada Cliff and Canyon	
CES206.902	Klamath-Siskyou Cliff and Outcrop	
CES206.903	Central California Coast Ranges Cliff and Canyon	
CES206.904	Southern California Coast Ranges Cliff and Canyon	
CES206.905	Mediterranean California Serpentine Barrens	
CES206.906	Mediterranean California Coastal Bluff	Coastal scrub (in part)
CES206.907	Mediterranean California Northern Coastal Dunes	Coastal scrub (in part)
CES206.908	Mediterranean California Southern Coastal Dunes	Coastal scrub (in part)