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A Review of Criteria for Evaluating Natural Areas

PAUL G. R. SMITH
JOHN B. THEBERGE
Faculty of Environmental Studies
University of Waterloo
Waterloo, Ontario N2L 3G1, Canada

ABSTRACT / Methods for evaluating natural areas have evolved in the last couple of decades to assess the importance of natural areas for the purposes of land-use planning, environmental impact assessment, and planning protected areas. Criteria used for evaluation vary and generally fall into three categories: ecological, or abiotic and biotic; cultural; and planning and management. Abiotic and biotic criteria are reviewed here in terms of three questions for each criterion: What is it—what are the definitions used in the ecological and environmental management literature? Why use it—what are the reasons behind its use? How has it been used—what is the state-of-the-art in assessing the criterion? Cultural criteria are discussed more generally in terms of the commonly used frameworks and the concept of significance. Planning and management criteria are generally related to either the need for management action or feasibility of effective management.

The idea of evaluating the significance of natural areas for conservation or other purposes has recently evolved and has found wide application (Margules and Usher 1981, Goldsmith 1983). The terms *ecological* and *conservation evaluation* have been used to describe the process (Ploeg and Vlijm 1978, Roome 1984). Such evaluations are used for environmental impact assessment, land-use planning, planning systems of protected areas, and management planning of individual protected areas.

Margules and Usher (1981) reviewed the criteria used in the evaluation of natural areas. The studies they reviewed were primarily on a local or regional scale in human-dominated landscapes. In this article we analyze criteria used on local, regional, national, and international scales, and in both highly developed and wilderness areas. International as well as Canadian and American experience is drawn on, and we include studies evaluating wetland, freshwater, and marine natural areas. Our purpose in doing this is to promote greater clarity in theoretical definition of criteria, emphasize the rationale for using particular criteria, and advocate better use of ecological theory and inventory data in criteria assessment.

The review presented here was conducted as a background for developing analytical tools to aid comprehensive planning of environmentally significant areas in the Northwest Territories of Canada (Nelson and others 1985, Smith and others 1986). The research is also part of ongoing research on environmental management in northern Canada (Theberge and others 1980, Fenge 1982, Bastedo and others 1984, Nelson and Jessen 1984).

KEY WORDS: Evaluation; Conservation; Criteria; Protected areas; Planning

The Development of Evaluation of Natural Areas

Ploeg and Vlijm (1978) discerned two types of ecological evaluation: "evaluation as an assessment of ecosystem qualities *per se* . . . regardless of their social interests" and "as a socio-economic procedure to estimate the function of the natural environment for human society." The differences between these two types of evaluations also represent a fundamental dichotomy in the rationale for conservation. On one hand "there are identifiable or potentially identifiable benefits to be derived through conservation . . ." and on the other hand "organisms have a right to exist" and "there is an undefinable though recognizable benefit to be derived from their mere existence" (Margules and Usher 1981). Livingston (1981) examined these two types of rationale for conservation and provocatively asserted that "there is no rational argument for wildlife preservation." Similarly, Ehrenfield (1976) identified a weakness in attempting to assign resource value to " 'useless' species or environments."

It is clear from the above that value judgments underlie both definitions of conservation and ecological evaluation and that the two definitions are not mutually exclusive. However, there are obvious differences between the evaluation systems to be reviewed here and systems which attempt to estimate the value of natural areas in monetary terms or other measures of social utility (Helliwell 1969, Langford and Cocheba 1978, Sinden and Windsor 1981, Sorg and Loomis 1985).

Most systems for evaluating natural areas use a series of criteria and, either quantitatively or qualitatively, evaluate each area's value with respect to each criterion. In another article we will discuss issues of measuring criteria and combining or otherwise using several criteria for indices of overall value. Here we

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Table 1. Criteria used in 22 selected evaluation systems.[a]

Criterion	Number of studies in which used	Type of criterion
Rarity, uniqueness	20	Biotic, abiotic
Diversity	20	Biotic, abiotic
Size	11	Biotic, abiotic, planning and management
Naturalness	10	Biotic, abiotic
Productivity	3	Biotic
Fragility	7	Biotic, abiotic
Representativeness, typicalness	8	Biotic, abiotic
Importance to wildlife, abundance	6	Biotic
Threat	6	Planning and management
Educational value	6	Cultural
Recorded history/research investment	6	Cultural
Scientific value	5	Cultural
Recreational value	5	Cultural
Level of significance	4	Planning and management
Consideration of buffers and boundaries	4	Planning and management
Ecological/geographical location	2	Planning and management
Accessibility	2	Planning and management
Conservation effectiveness	2	Planning and management
Cultural resources	2	Cultural
Shape	2	Planning and management, biotic

[a] Modified from Table 2 of Margules and Usher (1981) with 13 added studies (Nicholson 1968, Man and Biosphere Program 1974 and 1976, Ray 1975, Rabe and Savage 1979, Eagles 1980, Fuller 1980, Theberge and others 1980, Klopatek and others 1981, McCormick and others 1984, Canadian Department of Fisheries and Oceans 1982, Radforth and others 1981, McKinnon 1982, Parks Canada 1982). Only criteria used in more than one study are included in the table.

examine the types of criteria used in different evaluation systems.

The Criteria

A multitude of criteria have been used to identify and evaluate the significance of natural areas (Table 1). Names used for similar criteria vary from system to system. These criteria have been classified into generic classes of criteria adapted from Margules and Usher (1981). The criteria used in 22 evaluation systems are summarized in Table 1. Nine of these systems were treated in Margules and Usher's (1981) review. The precise terminology used by each of the 22 systems is given in Smith (1984).

The classes of criteria listed in Table 1 have been subdivided further according to their relation to abiotic, biotic, cultural, or planning and management considerations. Abiotic and biotic criteria relate to abiotic and biotic features and their characteristics per se and not to their potential for use by humans. However, as noted earlier, even these criteria involve human value judgments. Cultural criteria relate to cultural resources, particularly historical and archaeological sites, and to human uses of the landscape. Planning and management criteria relate to issues of importance from a planning and management perspective.

We will emphasize abiotic and biotic criteria here and discuss the others only in general terms. Each abiotic and biotic criterion is reviewed in terms of three questions: What is it—what is its meaning as presented in the ecological and conservation literature? Why use it—what is the rationale for its use in evaluation and its relation to other criteria? How has it been used—what is the method of assessment in various evaluation systems?

The order of discussion of abiotic and biotic criteria is based partly on Table 1 and partly on the use of ideas which relate to more than one criterion. A general discussion of cultural and planning and management criteria completes the analysis.

Abiotic and Biotic Criteria

Rarity

What is rarity? "Rarity . . . is based on geographic (restricted area) and demographic (low numbers) criteria. . . ." (Argus and White 1982). Preston (1948 and 1962) examined demographic rarity in quantitative

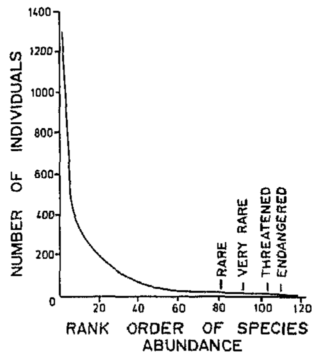

Figure 1. An example of the lognormal distribution showing the relative position of different categories of rarity. Based on Figure A from Eagles and McCauley (1982).

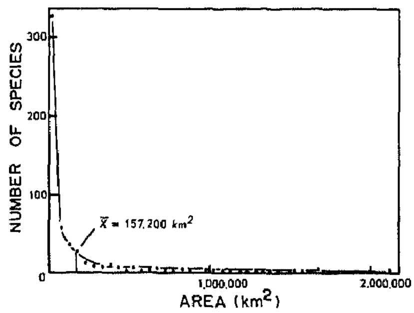

Figure 2. Range sizes of Central and North American mammal species. Endemic species stand on the *left of the mean* ($\bar{x}$) and the pandemic on the *right*. Redrawn from Rapoport (1982).

terms and noted that in samples of large numbers of individual organisms taken from biological communities the distribution of these individuals in the various species tend to form the characteristic mathematical distribution called the *lognormal* (Figure 1). In the lognormal distribution only a small number of species are very abundant while most other species are comparatively less abundant. Thus rarity, defined in terms of the lognormal distribution, is the relative position of a species on this curve (Figure 1, Eagles and McCauley 1982).

The concept of rarity also involves the spatial distribution of species abundance, which includes the size of species' geographical range as well as the patchiness in their distribution and abundance (Drury 1974 and 1980, Stebbins 1980, Harper 1981, Rabinowitz 1981). Geographic distribution can be documented more easily than abundance through atlasing projects and can provide a quantitative basis for defining rare species (Perring and Farell 1977). In many taxonomic groups the local abundance of species is positively correlated with the extent of the species' range (Brown 1984).

We discern five different types of rarity in the literature: widespread rare species "that occur over a wide geographical area but are scarce wherever they do occur" (Argus 1977) and may have a patchy or continuous distribution; endemic species with restricted geographical ranges; disjunct populations that are geographically separated from the main range of the species; peripheral populations that are at the edge of their species' geographical range; declining species that were once more abundant and/or widespread but are now depleted. The terms *endangered* and *threatened* represent extreme cases of any of the types of rarity noted. As well, most endangered or threatened species are declining or have declined.

Species with restricted ranges are described as endemic, but *endemic* is a relative term. An endemic species "only inhabits one place" and it "does not matter how big or small the place is" (Rapoport 1982:13). Figure 2 illustrates this point. Species to the extreme left of the diagram are extremely endemic and those to the far right are extremely pandemic ("inhabits all places").

Small populations of widespread species may be disjunct or isolated from the main distribution and thus have restricted range. These disjunct populations may differ genetically from the main populations of the species (Argus 1977). Additionally, they may often exhibit large fluctuations in numbers and have a high probability of local extinction (Soule 1973, Kilpatrick 1981). Disjunction is thought by many to be one important mechanism for genetic differentiation in the process of evolution (Endler 1977:6). "The primary effect of isolation (on the population) is a reduction in genetic diversity." The processes of population genetics which cause this reduction in genetic diversity (founder effect, genetic drift, bottlenecking), also cause the "occurrence of unique alleles and the increased frequencies of minor alleles" (Kilpatrick 1981).

Populations of species at the periphery of their range may have a limited distribution within a geographic region or political unit and thus be considered rare within that unit. For example, *Didelphis marsupialis* is widespread in the USA, but has an extremely restricted range within Canada. Such peripheral populations are also valued because they may possess unique genetic material. The same processes of population genetics that affect insular and disjunct populations occur in peripheral or marginal populations. Increased inbreeding, lower gene flow, genetic drift, narrower niches, and directional selection all influence marginal populations (Soule 1973, McClenaghan and Gaines 1981, Brussard 1984). All these processes lead to decreased genetic diversity, particularly inversion heterozygosity, within the population and greater genetic differences from central populations of the species (Soule 1973, McClenaghan and Gaines 1981, Brussard 1984). One example is the occurrence of "the most frost-hardy population of Douglas Fir, *Pseudotsuga mensiesii*, known in Canada" at the northern edge of its range in British Columbia (Argus 1977).

Why use rarity? The rationale for the conservation of all categories of rare species is the preservation of genetic diversity. The rationale is complex and has been stated in the World Conservation Strategy (IUCN 1980):

> The preservation of genetic diversity is both a matter of insurance and investment . . . and a matter of moral principle. The issue of moral principle relates particularly to species extinction. . . . We are morally obliged —to our descendants and to other creatures—to act prudently. Since our capacity to alter the course of evolution does not make us any less subject to it, wisdom also dictates that we be prudent. We cannot predict what species may become useful to us. For reasons of ethics and self-interest, therefore, we should not knowingly cause the extinction of a species.

How has rarity been used? Differing formulations of the notion of rarity were used in the evaluation systems reviewed. The variety resulted from considering different types of features and the different forms of rarity outlined above. Table 2 indicates the number of times the various types of rarity and different types of features have been explicitly used in the systems reviewed.

The method of assessing an area's significance with respect to rarity varies. Some systems assign a subjective score to each area based on the rarity of its biota and other features. One of these systems (Gehlbach 1975) arranged the different types of rarity in hierarchical order and assigned an arbitrary score to each: peripheral species, score = 1; rare, disjunct, and endemic species, score = 2; and endangered species, score = 3.

Assessments of rarity are often expressed as the number of rare species or features in an area. This necessitates the use of lists of species and features considered rare at one or a series of geographic scales, for example local, regional, and national scales (Ploeg and Vlijm 1978, Klopatek and others 1981, Dony and Denholm 1985). The number of rare species is sometimes correlated with the size of area, leading some to correct the rarity assessment for the effect of area (Dony and Denholm 1985, Miller and White 1986, but see Lahti and Ranta 1985).

In summary, rarity is a relative term and has many forms and consequently has been used in a variety of ways in evaluation systems. Its evaluation is particularly dependent on the existence of regional level information and syntheses of information.

Table 2. The number of times different types of rarity and different types of features were used in 11 evaluation systems.

Type of feature and type of rarity	Number of times used
Biotic—species	
Endangered or threatened	6
Rare	10
Endemic	3
Peripheral	2
Disjunct	4
Declining	1
Biotic—communities and habitats	
Rare	6
Declining	1
Rare in reserve system	2
Abiotic—Geomorphology	
Rare	4
Abiotic—aquatic features	
Rare	2

Diversity

What is diversity? According to Solomon (1979), "nowhere in the biological literature has a fundamental definition of diversity been given. . . ." Nevertheless, a generally accepted de facto definition has emerged: "the diversity of a community is . . . the number of its species (the community's species richness) and their relative abundance (called variously evenness, equitability or dominance). . . . The term species is to be interpreted broadly, including the usual taxonomic definition but also classifications based on other criteria . . ." (Solomon 1979). Thus, while the concept of diversity is most often applied to assemblages of species, the diversity of vegetation communities or geological features can be measured if a classification of the features exists so the different types can be counted (Pielou 1977, Romme and Knight 1982, Poleg and Vlijm 1978). Such classifications group all vegetation types, landforms, or other features which occur in a particular region into a series of categories that may or may not be organized hierarchically. The measurement of diversity has also been applied to different levels of taxonomic organization such as genera and families (Kaesler and others 1978, Salm 1984).

The most commonly used indices of diversity are the Shannon index, the Brillouin index, the Simpson index, and species richness (see Pielou 1977). Both Hill (1973) and Patil and Taillie (1979) show that these and other indices are related and can be specified by a single equation. By varying one parameter in the equation the different indices are obtained. The dif-

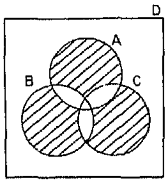

Figure 3. Diagrammatic representation of alpha, beta, and gamma diversity. The circles *A*, *B*, and *C* represent the alpha diversities of three areas; the *cross-hatched areas* represent the beta diversity of the three areas; and the square *D* represents the gamma diversity of the region which contains the three areas.

ferent indices vary primarily in how much weight is given to common and rare species (Hill 1973).

The number of species found in an assemblage varies with sample size, whether the sample is a collection of individual organisms or is a geographic area (Preston 1962, Simberloff 1978). To make valid comparisons between the diversity of different assemblages of species, the bias due to these two factors must be eliminated. Several techniques have been developed to correct for the number of individuals in the sample, with the rarefaction method being widely used (Simberloff 1978). The effect of area can be factored out using regression techniques (Connor and McCoy 1979, Dony and Denholm 1985). Ploeg and Vlijm (1978) stress that unless the effect of area on diversity is removed, the "criterion is completely unreliable for an objective comparison between areas."

Whittaker (1972) related the notion of diversity to geographic scale and spatial context by introducing the ideas of alpha, beta, and gamma diversity, which have gained wide usage in ecology. These are defined as follows: alpha diversity is the number of species in a particular site or area; beta diversity is the difference in species composition between different sites or areas; gamma diversity is the number of species which occur in any area or site within a particular region or landscape. Figure 3 graphically illustrates these three concepts. The geographical area to which gamma diversity is attributed is somewhat arbitrary. For example, the list of birds occurring in a region can be considered the region's gamma bird diversity. Alpha diversity has been emphasized in the evaluation of natural areas (Samson and Knopf 1982, Noss 1983). However, the idea of representativeness, particularly the "inclusive" definition, is related to gamma diversity. Classifications of natural diversity that are used in assessing representativeness, such as that of Radford and others (1981), are attempts to provide a framework for conserving a region's gamma diversity.

The notion of diversity can be applied to a variety of ecological features, although research has focused on species diversity. The measurement of diversity is well developed with a number of available methods which are appropriate for different purposes.

Why use diversity? The rationale for using diversity as a criterion is seldom stated explicitly in the conservation literature. However, several rationales exist, the most common being what might be called "more for your money." This rationale is stated here in relation to biosphere reserves: "It is desirable that a representative biosphere reserve should contain the maximum representation of ecosystems, communities and organisms characteristic of the biome" (Man and Biosphere Program 1974).

The presumed connection between diversity and stability is an often cited reason for using diversity as a criterion (see, for example, Eagles 1980). Margules and Usher (1981) contested this rationale, reasoning that stable ecosystems may in fact be less resilient and thus more at risk. Even more fundamentally, many ecologists contend that the long-held "notion that greater species diversity means greater functional complexity, in turn producing greater stability, appears to be no longer tenable, at least in any simple interpretation" (Colwell 1979; also see May 1973, Goodman 1975, Pimm 1984). Conversely, some recent studies (for example, Armstrong 1982) affirm MacArthur's (1955) conjecture that complexity may be positively related to some aspects of community stability. Much of this controversy is due to differing definitions of both complexity and stability as well as emphasis on different ecological variables (Connell and Sousa 1983, Pimm 1984; also see the section on *Fragility*, below).

Diversity often is linked to genetic variability (McKinnon 1982). That is, a large number of species in general possess a greater amount of genetic variation than a small number of species. Areas of high vegetation diversity in general provide the interspersion of habitats necessary for more life cycle components of more species than areas of more uniform vegetation (Environment Canada and Ontario Ministry of Natural Resources 1984).

How has diversity been used? "Species diversity for biological conservation is most conveniently assessed by measuring species richness. There seems little point in using indices which incorporate proportional abundance" (Margules and Usher 1981). The use of rarefaction (Simberloff 1978) or other complex techniques is not practical for ecological evaluation because the necessary data on abundance are often lacking. In the systems reviewed, diversity was often measured by ordinal ranking of sites and assigning subjective scores to the different levels. In all other cases richness was the

Table 3. The different types of diversity used in 11 evaluation systems.

Type of diversity	Number of times used
Biotic	
Vegetation communities	8
Habitats	3
Plant species	5
Animal species	
Birds	6
Mammals	5
Reptiles and amphibians	3
Fish	1
Invertebrates	2
Abiotic	
Geomorphology	3
Physiography	1
General	2
Aquatic ecosystems	1
All features	1

measure of diversity used. The diversity of different types of features was assessed—for example, biotic communities, species, abiotic features, and others. Table 3 lists the types of features considered in 11 studies and gives the number of times each was used in the evaluation systems.

Vegetation communities were most commonly used for assessing diversity. Probably this is because their inventory requires the least field work, and other forms of diversity are correlated with community diversity. To assess community diversity or diversity of abiotic features or aquatic ecosystems, a regional level classification is necessary unless subjective judgment is used. Plant, mammal, and bird species diversity were also commonly used where sufficient information existed. When good data for all species were not available, certain well-documented groups of species were selected as "indicators" for the assessment of biotic diversity (Ploeg and Vlijm 1978).

Many uses of the term *diversity* in evaluation fail to conform to any scientific defintion of the term. For example, Sargent and Brande (1976) define diversity in this way: "A natural area with a scenic view, such as a mountaintop, and with rare plants is rated higher than an identical area with no rare plants." Similarly, both, Gehlbach (1975) and Wright (1977) combined aspects of rarity and diversity in their indices of diversity. Wright (1977) defined his highest diversity category (score = 3) as: "Localized or relict community present or very good range of communities present."

The need to assess diversity relative to each area's size was mentioned earlier, and this has been considered in several evaluation systems (Ratcliffe 1977, Ploeg and Vlijm 1978, McKinnon 1982, Dony and Denholm 1985). Methods used in the Netherlands compare the diversity of a particular area to that expected for an average area of the same size using a standard species (diversity)–area curve (Ploeg and Vlijm 1978). Similarly, deviations from a diversity–area regression are sometimes used to assess relative diversity (Usher 1980, Dony and Denholm 1985). McKinnon (1982) used the mean number of tree species/0.5 ha as an index of an area's diversity. Ratcliffe (1977) took into acocunt the differing potentials among habitats for species diversity when making comparisons among areas. "Many more species of bird are likely to occur within a square kilometre of woodland than within a comparable area of upland" (Ratcliffe 1977).

Rudis and Ek (1981) incorporate the ideas of beta and gamma diversity into their mathematical programming model for selecting natural areas. Their model maximizes the gamma diversity or the total number of species contained in the system of areas selected. This is one of the few applications of beta and gamma diversity in evaluation of natural areas.

Size

Why use size? All rationales for valuing natural areas of large size make reference to the need to obtain large enough areas to capture and maintain the diversity of features, species, and genes in the regions where the areas exist and within individual areas. The application of the theory of equilibrium island biogeography (MacArthur and Wilson 1967) to the selection and design of nature reserves seeks to minimize the extinction of species in remnant patches of habitat in otherwise human-dominated landscapes. MacArthur and Wilson (1967) suggested the island analogy for continuous natural habitat that has been fragmented into habitat "islands." Diamond (1975) extended the analogy to suggest that the theory of island biogeography might be useful in designing systems of protected habitat islands. The equilibrium theory of island biogeography attempts to explain why islands often have fewer species than areas of equivalent size in continental areas (MacArthur and Wilson 1967). For islands, immigration rates may increase and extinction rate decrease with increasing size and decreasing degree of isolation, resulting in differing numbers of species at equilibrium depending on size and isolation (see Figure 4).

If we assume that (a) the theory is valid and (b) the analogy of nature reserves as islands holds because the land between reserves is so altered as to be a "sea" of inhospitable habitat, then a lower equilibrium number

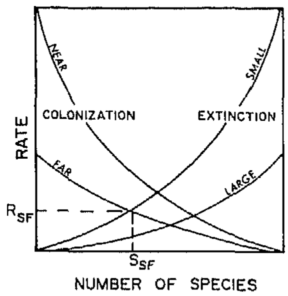

Figure 4. Extinction and colonization rates as a function of island size and isolation. R_{SF} shows the extinction and immigration rates for a small far island, and S_{SF} indicates the resultant equilibrium number of species.

of species would be expected for a small, isolated reserve than for a larger, less isolated one. Diamond (1975) translated these ideas into simple guidelines for nature reserve design.

Diamond's (1975) ideas have gained wide and often uncritical acceptance and use in the conservation literature (Goeden 1979, Miller and Harris 1977, Picton 1979, IUCN 1980.) However, the theory and empirical basis of Diamond's ideas have been hotly contested (Connor and McCoy 1979, Higgs 1981, Margules and others 1982, Simberloff and Abele 1982, Janzen 1983, Blouin and Connor 1985). Even the basis of the MacArthur-Wilson model of equilibrium biogeography is in question (Gilbert 1980, Sieb 1980, Busack and Hedges 1984).

A second rationale for large or minimum-sized areas arises from the fact that different species have differing range requirements and minimum viable population sizes (Soule 1980, Franklin 1980, Shaffer 1980, Schonewald-Cox and others 1983, Lehmkuhl 1984). "Ideally, reserves should contain populations of plants and animals which are both large and diverse enough to represent the genetic variability of those populations and to persist indefinitely" (Margules and Usher 1981). Shaffer (1980) advocates a focus on species at the top of the food web, reasoning that "if we are successful in providing sufficient room for their survival, then other, less space-demanding members of their communties should also survive." An example of this is attempts to calculate minimum size for a viable wolf population (Theberge 1983, Shields 1983).

Minimum viable population estimates focus on the "effective" population size, which may be considerably smaller than the censused population size, depending on sex ratio, population fluctuations, and variation in progeny number. To prevent loss of heterozygosity in small populations, some recommend that levels of inbreeding be kept below 1%. To meet this goal it is thought that effective population size should be at least 50 in the short term and 500 in the long term (Franklin 1980, Frankel and Soule 1981, Lehmkuhl 1984).

Lastly, it is generally considered desirable that reserves comprise units which are relatively independent of human activities outside their boundaries and associated impacts (Man and Biosphere Program 1974, Parks Canada 1982). "The ideal area is one which is sufficiently large to be self-regulating, through the inclusion of all the interacting components . . ." (Man and Biosphere Program 1974).

How has size been used? Size has been assessed in a variety of ways. Some systems consider the size of a potential reserve relative to a "minimum acceptable size" varying according to community type (Ratcliffe 1977, Tans 1974, Lovejoy and Oren 1981). For example, Ranney and others (1981) suggest that habitat islands of mesic forest less than 4–5 ha in size are almost entirely "edge" and thus not suited to conserve forest interior species. Size is often considered with other factors such as shape, location, and buffer zones to evaluate whether an area is an "effective conservation unit" (Man and Biosphere Program 1974, Tans 1974, Parks Canada 1982). In this case subjective scores are sometimes assigned (Tans 1974). McKinnon (1982) scored "management feasibility" based on size, shape, ecological integrity, boundary pressures, and existence of buffers. In some cases large areas are simply valued more highly than small ones: "the larger the area the more important it is for preservation in the public interest, other things being equal" (Sargent and Brandes 1978). In relating reserve size to its genetic resources, McKinnon (1982) scores area with a method which essentially converts area to number of species, using the slope of the species–area curve for Indonesian islands.

Naturalness

What is naturalness? "The term naturalness implies the recognition of some natural condition which may be difficult to determine. It is often used in a sense that implies freedom from human influence . . ." (Margules and Usher 1981). Such definitions of naturalness stress the absence of large-scale human modification. Many studies state that human uses should not be excluded if they are traditional and in harmony with the rest of the ecosystem (Ray 1975, Theberge and others 1980, Parks Canada 1982). Naturalness also implies the presence of natural regimes of disturbance such as windthrow, fire, and flooding (Pickett and Thompson 1978, Mooney and Godron 1983, Bonnicksen and Stone 1985).

The prevalence of the use of naturalness as a criterion is linked to its use in human-dominated landscapes such as Britain (Tubbs and Blackwood 1971, Ratcliffe 1977) and developed areas of the USA (Wisconsin—Tans 1974, Texas—Gehlbach 1975). The use of naturalness in these contexts has meant it is often negatively correlated with the occurrence of "alien" or introduced species (Ratcliffe 1977:7, Margules and Usher 1981). Thus, the "predominance of species widely accepted as native" appears to be a useful measure of naturalness in human-dominated landscapes (Margules and Usher 1981). Bonnicksen and Stone (1985) argue that naturalness of present vegetation should be established by comparison with quantitative descriptions of presettlement vegetation. This would involve comparison of the percentages of area covered by different vegetation types defined both by species composition and structure.

Why use naturalness? By far the most common rationale for the use of naturalness as a criterion is that undisturbed, natural areas provide the best source of baseline information to compare with other, considerably modified areas. Jenkins and Bedford (1973) summarized this idea well: "We have become uncomfortably aware that we do not even know how undisturbed ecosystems function, and that insight must be gained in this area before we can appreciate the true effects of ecosystem modification. . . . The more natural and protected . . . an area is, the better will it be suited for supplying baseline data."

Baseline areas must be both representative and natural, indicating a relationship between the two criteria in terms of human values. Furthermore, in human-dominated landscapes truly natural areas are exceedingly rare and are valued for this reason (Ratcliffe 1977:8, Adams and Rose 1978). Similarly, naturalness is also related to fragility in that the conservation of many fragile ecosystems and disturbance-sensitive species requires a great degree of naturalness or freedom from human disturbance.

Additional reasons for using naturalness as a criterion are tied to the many spiritual, philosophical, emotional, and recreational benefits often cited in support of the preservation of wilderness (Thoreau 1854, Leopold 1953, Livingston 1981).

How has naturalness been used? The methods used to evaluate naturalness vary, reflecting the scale at which the systems are used, the complexity of information, and the level of human disturbance within the region where the system is applied. Some methodologies used to evaluate small areas (Tans 1974, Gehlbach 1975, Wright 1977) simply classify areas into a few categories of naturalness and assign scores to these categories. For example, Wright (1977) uses the following categories and scores: "agricultural or artificial landscapes," score = 1; "seminatural landscape with native flora and fauna present," score = 2; and "near-natural landscape," score = 3.

The characteristics which indicate an area's naturalness vary from ecosystem to ecosystem, and thus the method of assessment must vary accordingly (Tans 1974, Ratcliffe 1977). For example, Ratcliffe (1977) evaluated naturalness of calcareous grasslands as: "lack of interference with physical structure (as by limestone mining and quarrying), and for relative freedom from sheep-grazing." For peatlands, however, naturalness was evaluated by assessing the degree of intactness of the mire structure, hydrology, and vegetation.

Studies by Parks Canada examine large areas composed of many ecosystems and must treat naturalness in a general way. For example, in one study, "twelve activities that might affect the 'naturalness' of an area were identified. The degree of impact was rated as none (2 points), light (1 point) or heavy (½ point). The points each area received were totaled and expressed as a percentage of 24 (no impacts at all)" (Inter-disciplinary Systems 1980). In general the naturalness criterion is treated as a constraint in selecting potential national parks. That is, some areas are screened out early in the selection process if they are "unnatural" (Inter-disciplinary Systems 1980).

Productivity

What is productivity? "Productivity is a measure of the rate at which communities of plants and animals bind energy into various kinds of organic material" (Peterson 1976). Productivity may refer to primary, secondary, or tertiary production, and it may measure gross or net values. The concept has been used with less confusion than other criteria for evaluating natural areas.

Why use productivity? The primary rationale for using productivity as a criterion is that areas of high productivity are unusual and often provide the energetic basis for production over a larger area. Larson (1982) explained that the most valuable wetlands along the Atlantic coast produce organic matter to fuel biological processes in adjacent waters. Peterson (1976) expressed a similar idea concerning areas of above-average productivity in the high arctic: "Meadow areas of relatively high productivity cover less than two percent of the Queen Elizabeth Islands but these habitats are of much more importance than their total area would indicate because they provide much of the primary productivity for the food web on the land."

Productivity is related to other criteria discussed here. The importance of an area to a species (to be discussed) is often related to the degree to which the area contributes to the species growth and reproduction. It has been claimed alternately that productivity is positively and negatively correlated with diversity (Connell and Orias 1974, Huston 1979).

How has productivity been used? Despite the fundamental importance of productivity, problems in its measurement and its correlation with other criteria have limited its use (Table 1). Carbon dioxide consumption, sequential harvest, as well as climatic and other environmental models are some methods used to estimate productivity in scientific studies (Bunt 1975, Hall and Moll 1975, Whittaker and Marks 1975). Remotely sensed data also show promise in estimating primary productivity (Botkin and others 1984). All such methods are time-consuming and expensive and thus are seldom used to evaluate natural areas.

The Canadian Department of Fisheries and Oceans (1982) used productivity as a criterion in developing a classification of aquatic habitats in northern Canada for planning hydrocarbon exploration. Oceanographic phenomena often associated with elevated levels of productivity were identified. These phenomena included polynias, ice edges, shore leads, and areas of upwelling, and were used in a qualitative way to place areas into four categories. An Ontario wetland evaluation model scores wetlands for productivity on the basis of growing degree-days, soil characteristics, wetland category, site location, and nutrient status (Environment Canada and Ontario Ministry of Natural Resources 1984). Similarly, Rabe (1984) used a sum of subjective scores for exposure, elevation, alkalinity, shallowness, bottom composition, and shoreline length to rate the productivity of alpine lakes. In that study the most and least productive lakes were considered most significant.

Oviatt and others (1977) showed that field measurements of productivity and other parameters can vary as much within one wetland as among different wetlands. This casts doubt on the usefulness of measurements of productivity and other criteria for comparative evaluation.

Fragility

What is fragility? The use of *fragility* and related terms such as *stability* has created semantic problems. Fragility and stability, whichever meanings are applied to them, are most often seen as opposite ends of a spectrum or gradient. Different populations, communities, ecosystems, and landscapes are positioned along this gradient.

Stability and fragility are complex concepts. Indeed, an ecosystem may be fragile according to one definition and stable according to another (Pimm 1984). For example, populations in a tropical forest may have low variability in numbers when disturbance is minimal but exhibit drastic declines or increases in the face of greater disturbance. In fact, the reason for the proliferation of terms and definitions has been to reduce this confusion by identifying different types of stability of interest (Harrison 1979, Sutherland 1981, Connell and Sousa 1983, Gigon 1983). Much of what has been written concerning stability is predicated on the existence of an equilibrium or series of equilibriums, which is in itself a controversial issue (Lewontin 1969, Westman 1978, Connell and Sousa 1983, Pimm 1984, Chesson and Case 1986).

Neighborhood, local or "Lyapunov" stability is most easily defined because a mathematical description of the phenomenon can be derived. This can be visualized as a marble in a bowl. At equilibrium, the marble is at the bottom of the bowl, and if the marble is displaced it will return to the equilibrium position. The rate at which the marble returns to the rest position at the bottom of the bowl is a function of how far it was displaced and the shape of the bowl. Outside the bowl, however, the marble will behave quite differently and there may be several locally stable points (Lewontin 1969, May 1973, Holling 1973). Specification of the global stability of ecological systems is far less tractable. Indeed, some would question the existence of global stability in any ecological system (Sutherland 1981).

In the recent literature there is a growing consensus on definitions of different aspects of stability. At the same time, little convergence exists in the terms used. The term *resistance* has been used to describe "the ability of a system to avoid displacement . . . during the stress period" (Harrison 1979, Pimm 1984), also called *inertia* (Orians 1974, Westman 1978). Pimm (1984) defined resistance more precisely as "the degree to which a variable is changed, following a perturbation." A related concept is that of persistence, the time a system remains unchanged during a perturbation (Connell and Sousa 1983, Pimm 1984). The speed with which a system returns to equilibrium has been called *resilience* (Harrison 1979, Pimm 1984), *stability* (Holling 1973), and *elasticity* (Orians 1974). The variability or constancy in the values taken by ecological variables through time is another stability concept (Orians 1974, Harrison 1979, Pimm 1984). A related notion is *amplitude* (Westman 1978) or the thresholds beyond which the values of variables will be permanently altered. Attempts have also been made to differentiate fragility due to natural and human-induced stresses and to further divide natural fragility into that due to external disturbances and fragility inherent in ecosystem structure (Gigon 1983).

All of the above ideas address some aspect of how ecologists perceive fragility and stability. In general, the use of the term *fragility* in evaluation of natural areas has referred to fragility susceptible to perturbation, most often human-induced.

Why use fragility? High fragility implies a high probability of "extinction" or "damage" of a species, feature, or system. Thus, the conservation of fragile ecological components requires protection from events, particularly human activities, which might cause "extinction" or "damage" (Ratcliffe 1977). Fragility is then often correlated with rarity, as rare items are thought to have higher probabilities of extinction or damage. Importance to wildlife also implies fragility. The more important one site is to a species the higher the probability that a catastrophic event could deplete the population or cause local extinction.

How has fragility been used? Usher (1980) stated that "the study of model populations has not yet advanced to a stage of being particularly useful in field assessments . . . [and] it is likely to be a long while before fragility can be assessed quantitatively." However, the conceptual and methodological basis now exists for measuring different aspects of fragility/stability. To date, little of this has found its way into conservation evaluation.

Wright (1977) ranked "sensitivity to disturbance" based on buffer area, size, and degree of threat. Sargent and Brandes (1978) ranked areas in Vermont based primarily on elevation, because areas at higher elevation are considered more fragile. Cairns and Dickson (1980) developed a method of estimating "ecosystem vulnerability" by drawing on several notions of fragility mentioned above and scoring each subjectively. Four components of vulnerability are defined:

1) Vulnerability to irreversible change
2) Degree of elasticity or ability to recover from damage
3) Inertia or ability to resist displacement of structural and functional characteristics (that is, the ability to resist being placed in disequilibrium
4) Resiliency or the number of times a system can snap back after displacement.

By far the most prevalent means of assessing fragility is in relation to a particular type of disturbance and the features thought to be most affected. Any analysis of such specific fragility is essentially risk estimation and is the realm of environmental impact assessment. A wealth of information can be used to assess such risks qualitatively. For example, there are methods to assess the "sensitivity" of terrain to vehicular traffic (Barnett and others 1977, Brown and Grave 1979), a method to evaluate the human, biological, and geological "sensitivities" of areas in the Beaufort Sea region of Canada to oil spills (Worbets 1979), and an "oil vulnerability index for marine oriented birds" (King and Sanger 1979).

Representativeness

What is representativeness? The representation in protected areas of the range of ecological variation is now a primary goal of conservation (see, for example, McNeeley and Miller 1984). The idea of representation is better thought of as an approach to conservation rather than simply a criterion.

There are two differing definitions of representativeness, which we will call *inclusive* and *typicalness*. The definition of Margules and Usher (1981) is inclusive: "Areas selected to be representative would necessarily include typical or common species but they could also include rare species since their objective is to represent the range of biota." This approach views the selection of reserves as a means to represent the full range of natural features in a system of reserves. A different view equates representativeness with typicalness: "Representativeness and uniqueness can be the extremes of a spectrum. A 'unique' area is one that is rare, whereas areas which are representative . . . are typical of a biome or habitat types. . . ." (Ray 1975). Usher (1980) grappled with a definition of typical: "By its definition a typical community . . . will contain all (or most) of the commoner and more widespread species. . . ." Typical areas might also be defined as having average values of diversity, productivity, and other attributes (Usher 1985, Rabe 1984).

The inclusive and typicalness concepts of representativeness are not necessarily mutually exclusive, and can in fact be complementary. For example, the methods of the Nature Conservancy Council (Ratcliffe 1977) sought to represent the "range of ecological variation" in a series of "key sites" but also used "typicalness" as a criterion for the selection of these sites. Similarly, the approach to representativeness taken by Parks Canada, detailed later, is intermediate between the two definitions. However, some have argued that areas with concentrations of "representative" natural features are atypical (Foresta 1985).

Why use representativeness? The rationale for using representativeness as an evaluation criterion depends to some extent on whether the typicalness or the inclusive definition is used. If the latter definition is used the rationales are similar to those given for conserva-

Table 4. Approaches taken to the selection of representative areas.

Location, scales, and reference	Classification used, means of assessment	Regionalization used and its basis
Great Britain 10^5 km^2 (Ratcliffe 1977)	Vegetation classification, professional judgment	None
North Carolina 10^5 km^2 (Radford and others 1981)	Classification of abiotic and biotic features, scoring system	None
Yukon 10^5 km^2 (Theberge and others 1980)	Professional judgment	Ecoregions *basis:* biotic (vegetation), climatic
Ontario 10^6 km^2 (Ontario 1978)	Classification of vegetation and abiotic features, professional judgment	Site regions, site districts *basis:* biotic (vegetation), climatic
United States 10^7 km^2 (US Park Service 1972)	Classification of biotic and abiotic features, scoring system	Physiography
Canada 10^7 km^2 (Parks Canada 1982)	Classification of biotic and abiotic features, scoring system	Terrestrial and marine natural regions *basis:* physiography and vegetation
World 10^9 km^2 (Man and Biosphere Program 1974)	Professional judgment	Biomes *basis:* biotic
World's oceans 10^{10} km^2 (Ray 1975)	None	Marine biotic provinces *basis:* biotic

tion in general. For example, "Who knows which component of our habitat is more significant than another. . . ? Who knows which elements of the biotic or abiotic habitat will be of great natural resource value to man? These fundamental questions can be answered only if we preserve total species/habitats diversity in carefully selected natural areas" (Radford and others 1981). Some rationales apply to both definitions: "the maintenance of large, heterogeneous gene pools; the perpetuation of samples of the full diversity of the world's plant and animal communities in outdoor laboratories for a wide variety of research; the protection in particular of samples of natural and semi-natural ecosystems for comparison with managed, utilized, and artificial ecosystems" (Nicholson 1968:15–16).

How has representativeness been used? Regardless of whether the inclusive or the typicalness definition of representativeness is used, two basic approaches to evaluating representativeness exist: a classification or listing of all natural features possible over a broad geographic region for comparison with those in candidate protected areas, or a regionalization or subdivision of a broad geographical region on the basis of biotic and/or abiotic features and subsequent selection of protected areas within each division. A particular evaluation system may incorporate both approaches. Table 4 shows how some systems which use representativeness use regionalizations and/or classifications of natural features. Selection of representative areas based on both regionalizations and classifications of natural diversity is common for national and provincial parks systems in Canada (Parks Canada 1972, Ontario 1978, Nova Scotia 1979, Quebec 1984), American federal protected areas programs (US National Parks Service 1972, MacFarland and Weinstein 1979, National Oceanic and Atmospheric Administration 1982), and the US National Wilderness Preservation System (Davis 1984). The variety of possible regionalizations and classifications of natural features is considerable even for a single geographic region (Bailey 1983, Iffrig and Bowles 1983, Clarke and Bell 1986). Two examples of how such classifications are used to assess representativeness will illustrate the process.

Both Parks Canada and the American National Parks Service use a regionalization of terrestrial areas based on physiography and forest types, and for each

Table 5. A weighting system for assessing representativeness.

Regional significance of natural feature	Value	Extent	Value	Representativeness value
Prime significance	2	Common	3	6
Prime significance	2	Uncommon	2	4
Some significance	1	Common	3	3
Some significance	1	Uncommon	2	2
Prime significance	2	Rare	1	2
Some significance	1	Rare	1	1

region develop a classification of "natural themes" or features which are representative of that region. Features in their classifications are scored in two ways: whether they are common, uncommon, or rare in each "natural region"; and whether they are of prime, some, little, or no significance or exceptional to a natural region (US National Parks Service 1972, Carruthers 1981). These scores are combined to give a value for the features in the natural region, as illustrated in Table 5. All the region's feature-values are summed, and this sum is used for comparison with sums obtained in exactly the same way for candidate protected areas in the region.

Note that the Parks Canada approach to the concept of representativeness is intermediate between the typicalness and inclusive concepts. The weights in Table 5 indicate that the common or typical features are emphasized but rare features are included with less weight.

The cooperative project of the Nordic countries, entitled "Representative types of nature in the Nordic countries," involves a particularly detailed and sophisticated approach to representativeness (Nordic Council of Ministers 1983). The Nordic countries were divided into 76 "physical–geographical regions" based on both abiotic and biotic attributes. A hierarchical classification of vegetation was developed proposing 600 types. A similar but less detailed approach was taken to landforms. A series of almost 400 "representative areas of nature" were selected by local environmental managers and inventoried, and the data were entered into a computerized data bank. These data provide a strong basis for assessing which vegetation and landform types are typical or rare as well as evaluating which sites are particularly representative.

Importance for Wildlife

What is importance for wildlife? One aspect of wildlife management is the identification of important habitat. Many adjectives have been applied to such habitat: sensitive, significant, critical, and unique are a few. The intent, however, is to identify areas that provide habitat for a certain, often large proportion of a wildlife population. Attention has focused on three components of the notion of importance: the relative importance of different species, for instance, from most endangered to most abundant and widespread; the relative importance of different components of the life cycle (breeding, migration, and so on) and the associated habitats; and the relative importance of the site populations, that is, the percentage of the total population considered important or critical. Depending on the definitions of these aspects of importance, the areas identified may be quite different.

The US Fish and Wildlife Service and National Marine Fisheries Service have associated the term *critical habitat* with endangered and threatened species: "'Critical habitat' for any endangered or threatened species could be the entire habitat or any portion thereof, if, and only if, any constituent element is necessary to the normal needs or survival of that species" (Baysinger 1980). Donihee and Gray (1982) state that in the Northwest Territories of Canada "management of critical habitats is necessary for habitats in short supply, or animal species in short supply, or habitats where animals are vulnerable to disturbance."

Why use importance to wildlife? There are many arguments for the conservation of areas important to wildlife. Some relate to the desire to prevent the extinction of species and the loss of genetic variability within species. These rationales have been discussed above in section *Why use rarity?* Another rationale is the management of wildlife populations for sustainable yield, which relates to the third basic goal of the World Conservation Strategy, the "sustainable utilization of species and ecosystems" (IUCN 1980). It is rationalized that the protection of "critical" areas will maintain "healthy" and dynamic populations.

How has importance for wildlife been used? A great many species occur within any geographic or political unit. Not all species are equally well suited to or well conserved by the protection of key areas. A variety of means have been used to determine the species of most concern.

Often no explicit method is used in selecting species of concern. Wildlife management has focused primarily on game species and others identified in legislation, such as migratory birds and rare, threatened, and endangered species. This focus has resulted in a species-oriented approach in which the identification of important or critical habitat is only part of a species management strategy. Nevertheless, the criteria for selecting species are sometimes explicitly stated: "The emphasis has been upon species that are, rare, threatened or endangered, and upon those that breed at traditional locations, in large concentrations, and in scarce habitat, or are particularly susceptible to disturbance" (Ealey 1981:36).

Adamus and Clough (1978) have identified those characteristics of species which make them more or less suitable and desirable for conservation in protected areas. Suitability criteria are site tenacity, seasonal mobility, area size needs, and spatial distribution. Criteria of desirability are relative scarcity, status changes, endemicity, peripherality, habitat specialization, habitat scarcity, susceptibility to disturbance, unique scientific values, and aesthetic amenities and use. Thompson (1978) developed a method using similar explicit criteria and subjective scores to assess what species are of most concern. He reasoned that "not all species are of equal concern; some have a relatively great importance or a high vulnerability" and it is necessary to determine "on a relative basis, which of the multitude of species found in an area are of greatest concern with respect to a particular land use decision." Scores assessed subjectively were assigned to each species for each criterion and summed, and this sum was used as a relative importance rating.

There have been few attempts to define general criteria to assess the relative importance of habitat used for different portions of the life cycle. Fuller (1980) indicated different abundance levels for breeding, wintering, and migrating populations which he considered significant at international, national, regional, county, and local levels. McCormick and others (1984) placed emphasis on sites with "populations which are concentrated, for any part of the year. . . . Such habitat sites include staging areas, moulting areas, nesting colonies, and the foraging areas of some species."

The variety of means used to assess the relative importance of site populations is, for the most part, a function of the quality of estimates of site abundance and whether or not accurate figures for the whole population exist. Smart (1976) recommended that a site supporting 1% or more of a populations of a waterbird population should be considered internationally significant. Good population and site abundance estimates are available for the United Kingdom and Eire, and thus Fuller (1980) and Lloyd (1984) were able to apply the 1% criterion. Fuller also applied the 1% criterion at other scales, national and regional, where data were available. Fuller also established numerical criteria for those species lacking total population estimates.

A series of techniques have recently been developed primarily in the USA to assess the importance of areas to wildlife species when site-specific abundance data are lacking and often not practically obtainable (see Sparrowe and Sparrowe 1978, Ellis and others 1979, Whelan and others 1979, and Seitz and others 1982 for reviews). These methods attempt to provide "professionally acceptable and reasonably uniform methods of judging the value of wildlife habitats" (Sparrowe and Sparrowe 1978). These methods are aimed at providing a tool for environmental impact assessment as they predict the longer term effects on habitat. Additionally, they furnish a means of analyzing tradeoffs of one tract for another. The models used are necessarily complex, and therefore we outline the conceptual basis of only one, the Habitat Evaluation Procedures (US Fish and Wildlife Service 1980).

Initially an area is delineated into cover types through airphoto interpretation. Target species are selected, and on the basis of existing literature, habitat variables most important for each species are chosen. The optimal value of these variables is assessed, and a function relating the habitat variable to "habitat value" is derived. Figure 5 portrays how average tree diameter of a cover type is converted to a habitat value index for white-tailed deer. For example, a forest stand with trees with an average diameter (DBH) of 10 cm is converted to an index of value to white-tailed deer of 0.2. Field sampling of the extant cover types is done, and the habitat variables identified above are measured. A "habitat suitability index" is then developed for each cover type, based on its level of achievement of optimal "habitat values." Then for each species, "habitat units" are calculated by multiplying the habitat suitability index for a cover type by the area occupied by that cover type. A habitat suitability index, ideally, is positively linearly related to the actual carrying capacity of the cover type.

Few tests of this ideal relation have been conducted, but one, by Binns (1978), found a very high correlation (r = .98) between a "habitat quality index" for trout and actual trout production. Such tests of "whether a model accurately predicts habitat quality" remain rare (Lancia and others 1982). Progress is being made in the use of satellite imagery in habitat

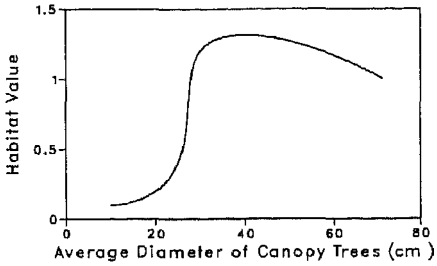

Figure 5. Illustration of a habitat suitability index. The habitat variable tree diameter at breast height is converted to *habitat value*. Data from Thompson (1978).

assessment models for some species (for example, Saxon 1983).

Cultural Criteria

Archaeological and Historical Features

Concurrent with development of evaluation for nature conservation there has been a growing need for evaluation of historical and archaeological sites for cultural resource management (Schiffer and Gummerman 1977, Sharrock and Grayson 1979, Tainter and Lucas 1983). Few of the evaluation systems listed in Table 1 assessed the relative significance of archaeological and historical resources (hereafter referred to as *cultural resources*). The purposes of those systems were primarily to identify and evaluate areas of biophysical significance.

An example of the types of significance often assessed for cultural resources are those of Schiffer and Gummerman (1977) and Schiffer and House (1977):

Scientific significance—whether a site's "further study may be expected to help answer current research question"

Historical significance—whether a site possesses good examples of resources characteristic of a particular "prehistoric culture, historic tribe, period of time, or category of human activity" and generally involves a cultural resource classification

Ethnic significance—"religious, mythological, social, or other special importance for a discrete population"

Public significance—value for public education and tourism

Legal significance—fulfillment of criteria defined by legislation.

Other evaluations have focused on the monetary value of artifacts, uniqueness concepts such as "earliest" or "biggest," and the conservation of representative samples of the main varieties of sites (Rabb and Klinger 1977, Glassow 1977).

All these frameworks have been criticized because assessment "is subject to variation between individuals, and to change through time" (Tainter and Lucas 1983). Tainter and Lucas conclude that "significance . . . is a quality that we assign to a cultural resource based on the theoretical framework within which we happen to be thinking" and as a result "objective significance evaluation is a myth."

Human Use

The evaluation of the significance of areas for different types of human use is a very broad field and relates to a large part of the land-use planning literature (for example, see Steiner 1983). Such evaluations include not only capabilities for different uses but past, present, and projected land uses and the compatibility of these uses and capabilities.

A large percentage of the evaluation systems reviewed use scientific, educational, and recreational use or potential as a criterion (Table 1). Some assign subjective scores (Tans 1974, Gehlbach 1975, Wright 1977, McKinnon 1982), as we have seen for other criteria. An example of this is Wright's (1977) scoring of potential educational use: "very limited school use or . . . suitable for individual or small group work," score = 1; "good use at most levels," score = 2; and "outstanding use at all levels of education," score = 3.

Several types of scientific significance have been used. Research potential is one type and is the same as that noted above in reference to cultural resources. A site is significant if its use may be expected to answer current research questions. In general, scientific significance is related to the identified need to have representative natural areas to provide baseline data for monitoring the effects of human activities. Conversely, some International Biological Program sites were selected as having "scientific interest because of the human management (alteration) to which they have been subjected" (Nicholson 1968). A second type of value is that of past scientific research or the "research investment" in a particular site (McKinnon 1982, Nicholson 1968, Ray 1975, Ratcliffe 1977). Lastly, many contend that most aspects of potential value for research are assessed by the ecological criteria (Booth and Sinker 1979, Moore 1982).

Planning and Management Criteria

Once the significance of an area has been established by ecological and cultural criteria, its importance must be assessed from a planning and management standpoint. Criteria for the latter assessment in general can be divided into two classes: need and feasibility. These considerations are often integrated into the decision-making process in an informal way but were sometimes explicitly treated in the evaluation systems reviewed.

A great many systems use need or threat as a criterion, some linking it to ecological fragility (Ray 1975, McKinnon 1982), others considering only the potential development or land-use and management response (Tans 1974, Gehlbach 1975, IUCN 1984). Need consists of a number of components. Threat of undesirable change is often used to set short-term priorities. Severity and imminence of threat are the aspects most often considered (for example, see Tans 1974, IUCN 1984). Replaceability or the availability of similar alternative sites is sometimes assessed (Quebec 1984). Long-term needs and priorities are more often established by relating management goals to the available ecological resources (Ontario 1981). For example, the management goal of one protected area in each biogeographic region can be used to set long-term priorities for site designation.

Feasibility criteria are used to assess the practicality of designating and managing an area. Consideration of feasibility is linked to the agencies conducting the evaluation and their mandates and power. Assessment done by an agency with a mandate for comprehensive environmental management will differ considerably from assessment by an agency with a more limited mandate. Nevertheless, there are many common concerns. Ecological, economic, social, and political factors are all important. Ecological integrity is often considered important for protected areas. They should be "ecological units whose long-term protection is feasible" (Parks Canada 1982), should be practical in size and shape, and should have "buffer potential or natural buffer" (McKinnon 1982). Social and political concerns necessitate "low land use conflict" (McKinnon 1982) and "minimum long term disruption of the social and economic life in the surrounding region" (Parks Canada 1982). Ease of designation and management includes factors such as local support and whether the site is on public land (Quebec 1980).

Concluding Remarks

Evaluation of natural areas involves both subjective and objective factors which are not easily separated (Roome 1984). Conservation is based on values and premises which are often inherent in the evaluation process (see Ploeg and Vlijm 1978, Ehrenfield 1976, Livingston 1981). For these reasons, we believe it imperative that criteria used be precisely defined, the reasons for their use stated explicitly, and the assumptions of any method of assessment be stated. Furthermore, each criterion should be related as much as possible to both the theory and empiricism of ecology.

Application of ecological principles to the evaluation of natural areas is a growing, dynamic, and controversial field necessitating that evaluation methods keep pace to ensure that decisions are based on sound principles.

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