Material for Breakout Session E:
Spatial Representation of Biodiversity and Conservation Values
and Ecological Services
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Spatial Representation of Biodiversity and Conservation Value

Description of method: This method results in the spatial representation of the uniqueness and irreplaceability of biological and ecological diversity in a regional context. This is a scientifically based approach to assign a conservation value to select species and ecological systems that are representative of an ecological region.

The values are represented as a numeric representation of the uniqueness, irreplaceability and level of imperilment for plant and animal species, vegetation, habitats and ecological systems.

Key assumptions:

- Representative biological and ecological diversity can be elaborated spatially across any region.
- The conservation value (status and quality) of each occurrence can be ascribed to each element of biodiversity as a repeatable and consistent procedure.
- The cumulative biological and ecological diversity and conservation values can be practically applied to inform and direct critical resource management and conservation decisions.

Key steps in the method include:

a) Define the biological and ecological targets for valuation
b) Define occurrence standards for each target
c) Define standards for valuing the quality of each occurrence
d) Define standards for measuring range wide status of each target
e) Create a ‘conservation value layer’ for each target that represents values and goals of the stakeholder

f) Create ‘conservation value summary’ of all targets that represents values and goals of the stakeholder

g) Modify the conservation value through incorporation of threats and opportunities.

Decision contexts where this method could be used:

- Enumeration of biodiversity protection implications that result from policy changes (i.e., change of protection status for isolated wetlands).
- Identification of critical riparian habitat
- Prioritization of remediation action on superfund sites
- Due diligence reviews and EIS as a prerequisite for permitting.
- Identification of reference conditions for establishment of baseline quality metrics for wetland and aquatic habitats.
- Assessment of the status of target species and ecosystems.

The method can be applied to a broad range of local to regional to national scales. The types of data and the spatial representation of this data change relative to the questions that are being addressed.

Resource inputs and limitations:

- The assumption is that there is a sufficient coverage of standardized biodiversity data required to run these models. The standards have been developed, and the data required changes associated with the application questions. Where there is a paucity of required data, it is readily ‘developable’, but can require the resources complete the required databases to run the models. The method is useless without good appropriate data.
- This method requires local scientific data, knowledgeable scientific interpretation and conservation planning expertise. The magnitude of the need is contingent upon the application and the current state of data and knowledge.
- Lack of data, currency and confidence of data, and data sharing issues associated with ‘sensitive’ data, training, and tools are the most important obstacles to the use of this method. However, there are many ways to create surrogate datasets that will allow users to adapt to different types of ‘barriers’.

Uncertainty. There are confidence measures built into the methodology that can be brought into the decision making process or displayed separately for analysis. The most significant sources of uncertainty in the use of this method include:

- The variability in the quantity and quality of the data.
- The limitations of scientific understanding of distribution and quality criteria for some elements of biodiversity.

Other important dimensions:

- The method is adaptable: it can be run repeatedly to represent temporal change or different landscape scenarios.
- Results are commonly aggregated to derive a single benefits number, but all of the native data is constantly maintained in the system and can be presented separately.
- The output is both understandable and communicable to the interested audience.
- The results are repeatable, and the process and algorithms are very transparent.

Detailed Description of Method

1. Define the biological and ecological targets for valuation

Biological diversity is often characterized by different levels of biological and ecological organization, from genes to populations to species to natural communities to ecosystems and sometimes to ecoregions and biogeographic provinces. All of these levels can be used for characterization and valuation, but certain levels are most appropriate to address specific types of assessments. For regional scale valuation, species, natural communities and ecosystems are generally used for purposes of conservation assessment and biodiversity valuation.

Within these categories, it is helpful to use the concept of coarse filter and fine filter conservation elements. The fine filter elements are important biodiversity resources that often are sparsely distributed across the landscape. These would include imperiled, declining, endemic, vulnerable, “umbrella” species and subspecies, as well as Focal Communities such as unique environments, rare plant communities, rare aquatic habitats, vulnerable species aggregations, migratory stopover points, and others. These fine filter elements represent those components of biodiversity that can become extinct due to lack of knowledge or attention. The coarse filter elements are comprised of the broad vegetation types, habitats and ecological systems that represent aggregations of communities and natural landscape patterns and processes at scales useful for management and monitoring. It is by looking at the combination of these fine and coarse filter element that one can portray the biological and ecological valuation of the landscape based on well developed and applied standards.

The valuation of fine and coarse filter elements across the landscape required a defined level for the currency and level of standardization of the knowledge. For example, there needs to be a defined taxonomy for all species and standard classification approach for all ecological units. NatureServe and the network of state Heritage Program currently maintain this level of currency and standardization for over 30,000 animal taxa, 56,000 plant taxa, 7,000 vegetation types and 1,500 ecological systems.

2. Define occurrence standards for each target

This methodology then applies the concept of recognizing an area of land and/or water in which a species or natural community is, or was present. These Element Occurrences (EOs) have practical conservation value for the Element as evidenced by potential continued presence and/or regular recurrence at a given location. Biologists and ecologists have developed criteria and have been conducting inventories for many decades to document the best occurrences of these elements of conservation across the landscape. NatureServe databases alone manage and distribute information on the occurrences of over 500,000 imperiled species across the United States. This number grows dramatically when adding freshwater and coastal habitats, vegetation types and ecological systems.

3. Define standards for valuing the quality of each occurrence

Each of the element occurrences defined above must be given a relative quality rank to allow planners, managers and conservations to prioritize their actions relative to management of the landscape. Biologists and ecologists have developed an approach to designate A, B, C, and D quality ranks to these fine and coarse filter occurrences of conservation elements.

These methods incorporate factors of occurrence size, condition and landscape integrity. Size factors that are used in this assessment include a quantitative measure of area of occupancy, population abundance, population density, and population fluctuation. Condition looks at biotic/abiotic factors, structures, processes within the occurrence as measured by population reproduction and health, development and maturity, ecological processes, species composition and biological structure, along with abiotic physical and chemical factors. Landscape integrity compiles a qualitative measure of biotic factors, abiotic factors, and processes surrounding the EO. These factors include landscape structure and extent, community development and maturity, intactness of ecological processes, species composition and biological structure, and additional abiotic physical and chemical factors.

Many of coarse and fine filter occurrence quality metrics have been developed and used to provide a quality/integrity attribute to all occurrences. The quality ranks portray what experts determine to be within acceptable ranges of variation. These ranges are developed through the characterization of multiple, apparently undisturbed examples, examination of impact and response to human-induced alterations, review of literature and historical records, and the development and testing of ecological simulation models. “A” ranked occurrences are within the preferred ecological integrity threshold. “B” ranked occurrences have one key factor within its acceptable range of variation. “C” ranked occurrences do not have any key factors with their acceptable range of variation, but they are still considered to be ‘restorable’. “D” ranked occurrences are no longer restorable. In some cases these factors can be directly measured, while in other they may be inferred/estimated indirectly.

4. Define standards for measuring range-wide status of each target

The next step in this approach is to assign a range-wide conservation status rank to each of the conservation elements. This is primarily completed and is most useful as an element attribute at the global scale, but the standards can also be applied at the national, sub-national and local scales. The conservation rank factors differ as they are applied to species as compared to ecological communities and habitats.

For **species**, the factors that are considered in assessing conservation status include total number and condition of occurrences (e.g., populations); population size; range extent and area of occupancy; short- and long-term trends in the above factors; scope, severity, and immediacy of threats; number of protected and managed occurrences; intrinsic vulnerability and environmental specificity.

For **ecological communities**, there are primary and secondary factors used in assessing conservation status. The primary factors for assessing community status are the total number of occurrences (e.g., forest stands) and the total acreage occupied by the community. The secondary factors for assessing community status are the geographic range over which the community occurs, long-term trends across this range, short-term trend (i.e., threats), degree of site/environmental specificity exhibited by the community, and the imperilment or rarity across the range as indicated by sub-national ranks assigned by local natural heritage programs.

The definitions for each of the Global (G) Ranks are:

G1 – Critically imperiled: At very high risk of extinction due to extreme rarity (often 5 or fewer populations), 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 – Uncommon but apparently secure: Uncommon but not rare; some cause for long-term concern due to declines or other factors
G5 – Widespread, abundant and secure: Common; widespread and abundant

All fine and coarse filter conservation elements across North America have been evaluated and given a conservation status rank.

5. Create a ‘conservation value layer’ for each target that represents values and goals of the stakeholder

The biodiversity value attributes that have been created for the global range-wide conservation status and the quality of viable occurrences now allows the development of a conservation value surface layer for each individual conservation element. The creation of this layer requires the ability to spatially portray each of the occurrences as well as the quality and confidence of each occurrence. The spatial portrayal of element occurrences

is derived from imagery, maps and field points, along with modeled distributions of specific elements. The element quality attributes are imported directly as available, and generated from landscape integrity models when necessary.

6. Create 'conservation value summary' of all targets that represents values and goals of the stakeholder

The combination of 'conservation value layers' for selected elements across a planning or assessment jurisdiction creates an aggregated 'conservation value summary' that provides a spatially explicit representation of the biodiversity and conservation values that are important to the conservation and resource management community. Different user groups can select the types of elements that there need to assess across the jurisdiction, and they can also modify the relative conservation weight of each fine and coarse filter conservation element. This will provide a customized conservation surface that portrays the values that they will need to incorporate into their planning and assessment work. This also becomes a baseline for monitoring the effects of their programs to manage for biodiversity value over time.

7. Modify the conservation value through incorporation of threats and opportunities in order to prioritize conservation and resource management activities.

The conservation values that are generated through processes 1-6 can be modified to reflect values that are relevant to a specific assessment. Zoning policies, growth models, economic values, ecological services and other values help to identify the effect of different or future scenarios relative to the current or desired future condition of the landscape.

- **Key Citations**

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Ecosystem Benefit Indicators (Boyd And Banzhaf, 2005, Banzhaf and Boyd, 2005, Boyd and Wainger 2002; Boyd 2004)

Because many ecosystem services are public goods, markets are not available to provide clear units of account. Cost-benefit analysis and national accounting for marketed goods is made easier by clear units of account: namely, the end-products consumers enjoy. While environmental economics has grappled for decades with the challenge of missing prices for environmental goods and services, it has neglected another central issue: the consistent definition of the environmental units to which prices are attached. An argument for standard units of account is that they can facilitate the transfer of valuations across the landscape and across time.

Possible use by EPA

Clear units of account are also desirable from the standpoint of environmental programs that police gains and losses in environmental quality or economic value. Consider wetland banking, water permit trading, land swaps, and natural resource damage assessment. All such activities trade compound, bundled environmental goods. Ideally, however, what should be traded – and accounted for – are the individual environmental goods and services provided by the bundle. In practice, however, trade and compensation programs use blunter proxies, such as "acres of wetland" or "pounds of nitrogen." What is lost in this kind of accounting system is gains and losses in individual ecosystem services.

Standardized units of account are also important to the measurement of performance. If the nation's environmental status is to be characterized and tracked over time units must be clearly defined, defensible ecologically and economically, and consistently measured. At present, the government and the public are presented with an over-abundance of units of measurement and often those units are poorly defined, unclear in their origin, and exacerbate the divide between economic and ecological analysis. Often within a single agency there are multiple competing paradigms for what should be measured.

At the national level, in the evaluation of new rules as part of the RIA process, government performance reviews, strategic planning, budget justification, and priority setting. They are also applicable at more local scales as a tool to improve regional and local planning, such as watershed planning in the context of TMDLs.

Description of the Method: Units of Account

There are two principle activities associated with the method. First, the definition and measurement of ecosystem service *units* (quantity measures of services). Second, the use of benefit indicators (or "willingness to pay indicators" to facilitate transfer of benefit

estimates across the landscape or to empower tradeoff analysis by regulators, planners, and conservancies.

Analysis of the benefits of natural resources requires a distinction between ecosystem components, processes, functions and *services.* The term services originates in economics, but has been adopted within ecology as well to signify the connection between ecosystems and human wellbeing.[1] Ecosystem components include resources such as surface water, oceans, vegetation types, and species. Ecosystem processes and functions are the biological, chemical, and physical interactions associated with ecosystems. These functions are the things described by biology, atmospheric science, hydrology, and so on.

Ecosystem services arise from these components and functions but are different: *Ecosystem services are the end products of nature that yield human wellbeing.* Part of this definition is particularly important: namely, that ecosystem services are "end products." End products are the environmental components about which people make choices. It is important to emphasize that many aspects of nature are *valuable*, but are not capable of being valued in an economic sense – because they are not associated with social or individual choices.[2]

This definition restricts the units of account, relative to many ways in which ecosystem services are commonly used. For example, nutrient cycling is often termed n ecosystem service. This is not a service, however, but rather an ecological function. To be sure, it is a valuable function, but it an intermediate aspect of the ecosystem and not an end product. Being valuable is not the same thing as being a service.

Consider another example. Reference is often made to recreation being an ecosystem service. It is not. Recreation is a benefit that relies on ecosystem services as inputs. Recreation is the joint product of ecosystem services including surface waters and fish populations and *other goods and services* including tackle, boats, time allocation, and access. From an economic standpoint, units of ecosystem account will exclude many things that are called ecosystem services.

Note that the above examples of economically defined units of account lead to units that are in fact biophysical, rather than "economic" in nature. An economic definition therefore leads naturally and necessarily to a bridge between economic and biophysical analysis. No ecologist should think that the economic definition of services leads away from biophysical analysis. In fact, the opposite is true.

The relationship of units of account to "environmental indicators" is as follows. First, the units of account described above are themselves indicators of performance or environmental conditions. These units are countable, spatially explicit indicators of certain biophysical characteristics. They can be expressed both numerically and spatially

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[1] See Gretchen Daily, *Nature's Services.*

[2] Many components of an ecosystem can be thought of as "intermediate products" in that they are necessary to the production of services, but are not services themselves.

via geospatial information systems. Thus, our units of account, or ecosystem service indicators, are related to certain "ecological indicators" emanating from the biophysical sciences.

Description of the Method: Willingness to Pay Indicators

However, we will also relate units of account to a different type of indicator: indicators of willingness to pay. In accounting for conventional, market goods, market prices are used to "weight" units of account. Because many ecosystem services lack these prices, how are units of account to be weighted? This question is central to benefit-cost analysis and welfare accounting. It should also, arguably, be central to government performance assessment and the evaluation of environmental trades, though preservation or enhancement of economic value is not always the aim of such programs. The aspiration of economic analysis is willingness to pay-based weights. For this reason, the workshop will also address the derivation of weights that can be assigned to ecosystem units of account.

The principle observation here is that the value of ecosystem services is highly dependent upon location in the biophysical and social landscape. In conventional accounting, arbitrage allows us to assume a single market price. For many ecosystem services there is no arbitrage. Also, many ecological services are best thought of as differentiated goods with important place-based quality differences. Ecosystem services' scarcity, substitutes, and complements are likewise spatially differentiated.

There are several implications. First, units of account should be spatially explicit. Second, the weights assigned to units – if units are to be aggregated into summary measures – should be spatially explicit. This can mean several things, depending upon the valuation method being applied. For example, stated preference techniques can be used to place value on units of account using place-specific scenarios. In other words, the scenarios presented in stated preference surveys could rely on standardized units and ways of measuring place-based quality, substitution, and complementary asset landscape factors. Alternatively, meta-analysis of existing value estimates can be used to calibrate benefit transfers. Standardized service units and location-specific factors affecting willingness to pay would provide a consistent architecture for such an exercise.[3] An alternative approach is a reduced-form regression of willingness to pay on various factors, including landscape-dependent indicators of the contribution of ecosystems to final goods and services and landscape-dependent indicators of substitutes and complements, population, and other socio-demographic characteristics.[4]

[3] This topic was raised at NCEE's workshop on benefit transfer in Spring, 2005.

[4] Willingness to pay, while not directly observable, is a function of various characteristics that are observable. WTP weights p_i can be thought of as a function of landscape indicators I. In principle, this function, on a service-by-service basis, can be calibrated by relating observable indicators I to existing WTP estimates of service value. Were this possible in practice, location and ecosystem-specific indicators I could be used to transfer monetary WTP estimates to locations where they are not available.

Finally, there is relevance to less econometrically formal weighting procedures. Examples here include stakeholder-driven decisions, citizen juries, and mediated modeling exercises. In these examples weights are not derived by economic analysis, but rather are debated and concluded via some kind of institutional process. Here too, standardized units of account and landscape willingness to pay indicators could help educate and discipline benefit assessment.

Strengths and Weaknesses

Both ecosystem service measurements (indicators) and benefit indicators are a quantitative and visual, but not monetary, approach to the assessment of services. Unless married to an econometric benefit transfer exercise, conjoint analysis, citizen jury, or other weighting approach, the indicators will not themselves yield a single dollar-based "answer." Rather, they should be though of as an accounting tool to measure and track over time, in a consistent manner, changes in service levels and factors related to willingness to pay for those services. The monetization of benefits, which is clearly important in certain regulatory applications, demands additional methods.

Service and benefit indicators are simple, countable aspects of the biophysical and social environment. They are transparent and easily replicable. Because indicators are cheaper to generate than econometric value estimates they better allow for landscape assessment of multiple services at large scales.

EBIs are drawn mainly from geospatial data, including satellite imagery. Data can come from state, county, and regional growth, land-use, or transportation plans; federal and state environmental agencies; private conservancies and nonprofits; and the U.S. Census. Benefit indicators can capture the landscape, or spatial, factors that contribute to social well-being. This is in fact a virtue of indicator methods. Indicators can be derived from and mapped within a GIS context. Spatial analysis is important because the ecological production function is a function of spatial interdependencies. From an economic standpoint, the social determinants of service benefits depend upon the landscape context in which those services arise. The consumption of services often occurs over a wide scale. Habitat support for recreational and commercial species, water purification, flood damage reduction, crop pollination, and aesthetic enjoyment are all services typically enjoyed in a larger area surrounding the ecosystem in question.

The method is applicable to the full range of ecological services. In practice, applicability may be limited by data gaps.

The principle disadvantage of indicators alone, is that they do not directly yield dollar-based ecological benefit estimates. They also do not in themselves weight or estimate the tradeoffs associated with different factors relating to benefits (though as noted above they can be married to more formal methods designed to do such weighting). This is not really a weakness to indicators themselves, but rather an acknowledgement that more must occur than simple indicator measurement if the goal is dollar-based end-results.

Uncertainties associated with the method and how they would be addressed: A core rationale for the use of a benefit indicator approach is to explicitly convey the sources of complexity – and hence uncertainty – characterizing biophysical systems and the service flows arising from them. The visual depiction of willingness to pay indicators, for example, can mimic sensitivity analysis by presenting a range of benefit scenarios in GIS form. However, the visual depiction of quantitative information introduces uncertainties of its own. In particular, visual depictions can strongly influence perceptions. Uncertainty with regard to how indicators are perceived, particularly when presented visually should be acknowledged.

- **Key Citations**

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