Institute for Water Resources, Policy Studies Program

# Improving Environmental Benefits Analysis In Ecosystem Restoration Planning


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*This report contains discussion about analytical approaches and procedures that are not yet approved by HQUSACE for broad Corps-wide use. Contact HQUSACE for specific guidance before applying the procedures proposed in this report.*

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*http://www.iwr.usace.army.mil/iwr/ecosystem/envirobenefits.htm*
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# Executive Summary

This study examined ecological and economic concepts relevant to improving environmental benefits analysis, and recommends a strategy for improving related tools, application capabilities, policy and guidance. In planning ecosystem restoration projects, the Corps uses non-monetary indicators of benefit in cost-effectiveness analysis and incremental analysis, rather than economic benefit-cost analysis. These non-monetary indicators can be supplemented by consideration of incidental monetary benefits. In cases of joint formulation for contributions to both National Ecosystem Restoration (NER) and National Economic Development (NED), formulation and evaluation uses a combination of monetized NED outputs and non-monetized NER outputs. Ideally, all significant effects from projects would be expressed in the same unit-measure, but technical limitations currently impede this. Analytical difficulty escalates as the number of outputs considered increases, thus, a significant challenge in ecosystem restoration planning is reducing the number of different outputs considered in cost-effectiveness analysis to a manageable number of those most significant.

The study confirmed what Corps policy has concluded—that planning principles for NER and NED objectives are essentially the same, except for "limitations in understanding the complex inter-relationships of the components of ecological resources and services", and except that "the environmental outputs...are typically not monetized." Two different premises for guiding pursuit of ecosystem restoration were found. One is skeptical of human management toward specified ends, favoring instead to simply remove the sources of human effect, then, let nature take its course, whatever that may be. Implicit is value on the unspecified range of natural outcomes that results. The other premise places more faith in human enterprise, emphasizing benefits from a set of resource services using habitat improvement measures that simulate nature (the naturalistic approach) as well as natural features and processes. Corps policy includes elements of both premises, but does not clearly address interrelationships of naturalness, resources and services – appearing to emphasize restoration to a more natural condition, but urging examination of significant outputs in evaluation.

The study reviewed existing concepts of ecological resource and service flows, including numerous interrelationships regarding ecosystem structure, function, integrity, sustainability, health, resilience, functional stability, and biodiversity. Of those, biodiversity proved to be most inclusive concept underlying indicators of ecosystem naturalness and human effect, once calibrated against an array of reference conditions with varying degrees of human effect. The study revealed no widely applicable environmental (non-monetary) indicator of human benefit. However, a "biodiversity scarcity index", based on the uniqueness and vulnerability of species and their associated genetic information, deserves further consideration as an inclusive indicator of NER contributions. No ideal models and methods for ecosystem restoration planning exist at this time, but several types of biodiversity-based models and methods can be used cautiously in the interim, even though they have substantial limitations. Only rarely will an existing single model be comprehensive enough for complete evaluation, especially when the resources recognized for their social significance comprise a subset of the biotic community. Despite these limitations, many advances in recent years now allow development of much improved models.

The study also concludes that no fully inclusive monetary measure of ecosystem restoration effects now exists, and that significant technical obstacles currently preclude the economic valuation of all restoration outcomes. This suggests that the current policy guidance that recognizes non-monetary NER outcomes as a category of effects separate from monetary effects will continue to be appropriate for the foreseeable future. The cost-effectiveness analytical framework is very useful, even when outputs are expressed in multiple, non-commensurate metrics, including joint NER/NED planning. However, analytical complexity increases as the number of non-commensurate metrics increases beyond two or three, making pursuit of inclusive metrics important. A proposed strategy focuses on this pursuit through coordinated improvement of techniques, policy and guidance, and practitioner capabilities.

The proposed strategy involves overlapping (I) *near-term, (II) intermediate,* and *(III) long-term* components, all of which would start immediately. The *near-term* or **Incremental** component (2 to 3 year horizon), addresses the requirements of the current Corps planning regulations and emphasizes broadened staff proficiency and selection and application of existing ecological assessment models. Elements include:

- Greater use of community-habitat models instead of emphasis on single-species habitat models.
- Improving guidance on model selection and use.
- Incorporating the concepts of and approaches to environmental benefits analysis into courses, workshops and other forums designed to enhance staff capabilities.
- Refining policy and guidance pertaining to ecological and methodological concepts and the concept of NER.

The *intermediate*, or **Next Generation** component (5-year horizon), pursues a fundamental rethinking of the NER objective and desired outputs, including possible specification of ecological services in the definition of an NER account, improvement of multiple-output evaluation models based on simulation of geophysical and ecological process understanding, and exploration of new analytical frameworks for multipurpose NED/NER planning. Elements include:

- Developing and refining multi-output ecosystem simulation models consistent with improvements in Corps plan evaluation frameworks.
- Further investigating and developing a metric based on the scarcity of biodiversity resources to indicate NER contributions.
- Refining baseline and strategic R&D programs to include wider applications of existing planning models and development of appropriate new models.
- Refining the NER concept and analytical frameworks for joint NED/NER projects relative to the concept of ecosystem services, sustainability, and the “NEPA process” in pursuit of more consistent evaluation standards.

Over a *longer-term* (10+ year horizon), efforts could be made to explore pursuit of economic valuation of ecosystem services. This **Monetization** component attempts to allow more NER outcomes to be evaluated in NED terms, thereby reducing the total number of non-commensurate choice criteria that would need to be considered for efficiency analysis. Elements include:

- Examining the technical obstacles including uncertainties in forecasting ecological outputs, and limitations in measuring non-marketed goods.
- Developing technically and politically acceptable estimates of monetary value for restoration effects when practicable for use in the CE/ICA framework.

# Report Summary

## Introduction

The U.S. Army Corps of Engineers (Corps) now pursues ecosystem restoration as a priority Civil Works purpose on par with traditional economic development purposes such as commercial navigation and flood damage reduction. But whereas the Corps planning framework applied to economic development purposes requires a monetary standard for project evaluation, the framework established for ecosystem restoration planning stipulates that restoration project outputs are to be measured and expressed in non-monetary metrics.

The recent emphasis on assuring returns from Federal investments, along with the reality of budgetary constraints, have resulted in a renewed interest in the methods used to evaluate the outputs of environmental projects and programs. Importantly for the Corps, the adequacy of the methods currently being used within the Civil Works program for characterizing and evaluating the environmental outputs of ecosystem restoration projects has been questioned in a number of forums (e.g. NRC (1999), OMB).

### Purpose

This report identifies and examines policy and technical issues related to improving environmental benefits analysis for Civil Works planning. As used here, the term "environmental benefits analysis" refers to the development of an evaluation philosophy, framework and complementary analytical tools to aid Corps restoration project planning, but is *not* intended to imply a planning process that involves estimating dollar values for restoration outputs.

The primary report focus relates to the identification and assessment of alternative metrics and analytical procedures for characterizing and evaluating restoration project outputs in non-monetary terms. A secondary focus relates to the identification and assessment of alternative plan comparison frameworks for projects plans formulated at least in part to serve ecosystem restoration. With regards to these issues, the report addresses the following questions:

- What non-monetary metrics of environmental quality change may have wide applicability for characterizing and evaluating ecosystem restoration outputs?

- What plan comparison frameworks and procedures are available for plans formulated to serve ecosystem restoration as well as mixed economic and environmental objectives, and what are their strengths and weaknesses for illuminating the economic efficiency implications and tradeoffs among plans?

## Civil Works Ecosystem Restoration: What, How and Why?

**Section 2** describes the planning framework established for civil works ecosystem restoration. It outlines the basic guidance established for civil works ecosystem restoration planning, and examines possible implications for characterizing and evaluating restoration outputs.

### Focus of Ecosystem Restoration

Corps environmental management expertise generally centers on the hydrology and geomorphology of aquatic systems. Corps restoration projects typically focus on significant water and related land resources of river and coastal ecosystems, including their associated floodplains, shores, and wetlands. The boundaries of these natural systems typically define the appropriate focus of all civil works activities, including traditional flood control and navigation projects as well as ecosystem restoration activities. But whereas traditional civil works projects generally rely on management measures to eliminate hydrologic extremes, ecosystem restoration generally requires measures to re-establish natural hydrologic variability.

### Objectives of Ecosystem Restoration

Civil works activities alter the structure and processes of ecosystems. The evaluation of such alterations for decision-making purposes requires a standard of value for measuring the outcome associated with such change. That valuation standard should follow logically from the stated objective in civil works planning.

In traditional civil works planning, the federal objective is defined as utilitarian; that is, to contribute to the satisfaction of human preferences. Economic value provides an empirical "account" of the contribution of civil works activities to preference satisfaction, and represents the standard of value specified by Corps guidance for the evaluation of traditional civil works projects. Further, Corps guidance specifies the specific purposes -- or desired outputs -- to be served by traditional civil works projects. These traditional outputs can be viewed in terms of closely related "ecosystem services".

Ecosystem services have been defined as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Dailey, 1997). As this definition implies, ecosystem services can be viewed as the link between the natural properties of ecosystems and human benefits. That is, the service concept connects an ecological focus on "what ecosystems do" with an economic focus on how ecosystems contribute to the satisfaction of human preferences. As such, the concept embodies both an ecological dimension and a human dimension. Box ES-1 provides a list of example ecosystem services and the various ways in which they can contribute to economic value.

Traditional civil works purposes include many of the production and consumption activities listed in the right hand side of Box ES-1, and these in turn are closely linked to one or more underlying ecosystem services listed on the left side of the table. Indeed, it is these associated ecosystem services that are the focus of plan formulation for traditional civil works projects. So, for example, commercial navigation projects focus on intensive enhancement of natural waterway transportation links, and flood damage reduction projects focus on enhancement of the natural flood regulation service.

<table>
<tr><th colspan="3">Box ES-1. Example Ecosystems Services and Associated Human Uses & Benefits</th></tr>
<tr><th>Ecosystem Services</th><th colspan="2">Channels Through Which Ecosystem Services Contribute to Economic Value</th></tr>
<tr><td rowspan="3">• Disturbance Regulation (flood, wind & wave)<br>• Waterway Transportation Links<br>• Water Storage<br>• Water Purification<br>• Sediment Trapping<br>• Waste Treatment<br>• Biological Pest Control<br>• Climate Regulation<br>• Rare and Unique Species/Genetic Store<br>• Wildlife Support (e.g., food chain, nursery)</td><td>Direct Passive Use</td><td>• Personal satisfaction derived from the knowledge that rare ecosystems & associated functions & services are intact, independent of any actual or anticipated active use</td></tr>
<tr><td>Direct Consumption</td><td>• Community Flood & Storm Protection<br>• Municipal & Residential Water Supply<br>• Consumptive & Non-consumptive Recreation<br>• Aesthetics, Observation & Study</td></tr>
<tr><td>Production Inputs</td><td>• Land Productivity for Agriculture<br>• Commercial Navigation<br>• Hydroelectric Power Generation<br>• Water Input for Agriculture & Industry<br>• Commercial Fishing, Hunting/Trapping, etc.</td></tr>
</table>

While the relationship between ecosystem services and the traditional civil works objective and specific purposes is straightforward, the relevance of services to the ecosystem restoration mission is not as apparent. Corps planning regulations and supporting policy documentation specify that the federal objective in ecosystem restoration is to “increase the net quantity and/or quality of desired resources” through the restoration of “significant ecosystem function, structure and dynamic processes that have been degraded”. The relevance of ecosystem services for the restoration mission depends on how this objective statement is interpreted in terms of desired ends. At least two possible motives for movement along a restoration path can be identified.

First, restoration might be sought purely for utilitarian reasons, implying a concern for services that people value. That is, management actions might seek to restore the hydrologic conditions thought necessary to secure a mix of ecosystem services and associated human benefits only because that is the best plan for reestablishing deficient services.  But, when services ordinarily associated with a more natural condition are better gained by artificial means, a simulation of natural measures—a naturalistic approach—or even a highly artificial plan, might be chosen. The approach to restoration might be called “the manager knows best” approach and is based on careful analysis of resource and service flow from a variety of natural and artificial management measures. Natural ecosystem services can produce economic value in a variety of ways. In the extreme, a person may gain individual utility from, and thus be willing to pay for, the mere knowledge that a resource and associated services are maintained in good condition. Such assurance is said to produce “passive use value” that is independent of current or future plans for use. By contrast, “use value” is generated when people actively use ecosystems services by consuming them directly or as inputs into commercial production. For example, restoration can augment water purification and wildlife support services that generate use value directly by improving recreation opportunities, and indirectly by supporting commercial fisheries. Restoration of nature's services can also generate use value in more subtle and indirect ways by supporting general economic and social activity. For example, services such as climate regulation, sediment trapping, and waste treatment support and prevent damage to a wide range of consumption and production activity.

A second possible motive for pursuing restoration is the "naturalness" of ecosystem hydrology and geomorphology, as an end in itself. This approach is not independent of the resulting mix of service flows, but assumes that whatever results ecologically is more acceptable than the results from any other alternative. This might be called a "nature knows best" approach. It ignores the service flows from proposed management measures based on the *a priori* judgment that no better plan alternative exists. On the surface at least, this seems to be the position of some environmental interest groups that advocate a return to free-flowing rivers in certain contexts. The notion that naturalness is an independent value to be advanced by civil works planning is at odds with the traditional civil works objective rooted in utilitarianism, but represents one plausible motivation for Corps restoration activities.

**Evaluation of Ecosystem Restoration**

Corps planning guidance does not specifically establish the goal of restoration as naturalness as end in itself, or as a means to support desired natural service outcomes. Instead, Corps regulations emphasize the "significance" of resources and effects for guiding restoration planning. The significance concept is defined in terms of institutional, public or technical recognition, and as such seems broad enough to include both naturalness and associated services as desired restoration ends.

Corps regulations specify that restoration outputs must be evaluated in non-monetary metrics, with preference given to "units that measure an increase in ecosystem value" that are indicative of the social significance of project effects. To the extent that significance in any particular restoration context relates directly to some natural ecosystem condition (e.g., a free-flowing riverine ecosystem), then metrics indicating a change towards that natural state may be all that is needed for project evaluation. On the other hand, if significance relates to specific ecosystem component parts or processes by virtue of their links to valued services (e.g., a particular fish species), then project evaluation may need to include metrics indicating the desired direction of change in these components.

**Ecological Concepts Underlying Environmental Benefits Analysis**

**Section 3** presents and discusses ecological concepts and theory relevant for restoration planning. It reviews the relevancy of ecological concepts to consideration of ecosystem-output indicators of ecological "value", and considers their potential to provide a basis for defining a single ecological metric for characterizing and evaluating restoration benefits.

Holistic views of natural ecosystem structure, function, and other processes are expressed through various concepts, including ecosystem integrity, biodiversity, self-regulation, resilience, stability, sustainability, production, and materials cycling, among others. Many questions remain about concept validity and practical application, but these concepts reveal some potential for application and many of the problems encountered in attempts to develop widely applicable indicators of environmental value.

**Structure, Function, and Service, and Their Sustainability**

The structural and functional properties of ecosystems define their component parts and processes, their diversity, their sustainability, and their response to influential forces originating from outside the ecosystem, such as solar radiation and gravity. Structure and function are characterized by biodiversity, which is the biologically derived variety that occurs in ecosystem form and dynamics. Genes are often said to be the most fundamental of ecosystem parts because they hold the information needed —the “architectural plans”—for sustaining all other ecosystem structure and function, and associated natural services. The genetic information in ecosystems is most typically indicated in the biodiversity expressed in species and communities. Thus biodiversity indicators of genetic scarcity, including its vulnerability to extinction, shows potential as an indicator of environmental resource value once their expression is matched with indicators of human service. A disproportionate amount of unique genetic information is sustained in relatively rare species. Biodiversity loss through human-caused acceleration of species extinction erodes ecosystem “design and maintenance” information held in genes, and the natural capacity of ecosystems to sustain the diverse structures and functions supporting all dependent ecosystem services and values. This erosion of biodiversity limits management options, precluding possibilities that are dependent on the extinct genes.

**Ecosystem Integrity and Biodiversity Measures of Naturalness**

Ecosystem integrity relates to the completeness of natural structure and function, usually as indicated by native biodiversity. Full integrity is exhibited under the conditions with which the ecosystem evolved, in the absence of modern human impact. In practice, the least modified of natural reference conditions is selected to anchor an index of ecosystem integrity in its maximum value. One of the purported hallmarks of natural integrity is the sustainability of natural ecosystem structure and function that results from self-regulation under evolutionary conditions. As natural integrity is lost, the remaining structure and function, and dependent natural services, tend to become less predictable. However, most ecosystem-level function and natural services associated with production, biomass and materials flow and recycling is associated with common species and erodes much more slowly than structure and functions that sustain unique genetic information.

Variation from natural integrity is determined by the observed biodiversity in an array of reference-ecosystem conditions approximating a range from fully natural to fully altered states. A multivariate measure of natural biodiversity is a more inclusive indicator of natural ecosystem integrity than a single-variable indicator such as number of species. In practice, the diversity of a species subset, such as fish or birds, is most commonly used to indicate ecosystem condition. Such measures can fall short of desired inclusiveness because they are incompletely representative of ecosystem habitat and community conditions. Less inclusive still are single-species indicators of ecosystem integrity.

**Concept of Integrity In A System Context**

Conceptual models of ecosystem integrity are least well developed at the landscape scale of structure and function important for recovery of local integrity loss following disturbance and sustaining ecosystem-wide integrity. Ignorance of the system context in the influential landscape distorts the evaluation of plan effectiveness in restoring and sustaining resource production. Local integrity can be restored, once lost, when all of the parts, processes, and recovery pathways connecting intact and locally disturbed areas are maintained. Ecosystem resilience—the capacity to "bounce back" from stressful disturbance—restores local integrity through habitat recolonization and community succession. Functional recovery ordinarily is more rapid and complete than structural recovery at a local site. Rare species colonize individual sites much less reliably than common species and their maintenance usually requires a larger area of intact natural integrity. Compared to common species, the colonization of specific sites by rare species appears to depend substantially more on random events.

**Cultural Integrity and Ecosystem Health**

The concept of natural ecosystem integrity emphasizes the connections among fully natural integrity, self-regulating functions, and sustainable states, but is less clear about relationships in less natural ecosystems. The concepts of "cultural integrity" and "ecosystem health" suggest that ecosystems can have numerous modified but sustainable states without being fully natural. Cultural integrity can exist locally and/or globally within the ecosystem. A locally modified portion of an ecosystem, say a waterway, can sustain function, once altered, albeit with a locally altered biodiversity. As more of the ecosystem is converted to human use, the natural attributes of ecosystems gradually erode and eventually threaten the sustainability of the rare and unique structure and function. A sustainable condition of cultural integrity and ecosystem health exists when enough natural integrity is maintained in the ecosystem to assure that recovery is possible anywhere in the ecosystem, if society so chooses. The most usual strategy used to restore cultural integrity and health locally is to improve habitat quality and connectivity within the physical limits of beneficial enhancements, such as improving clean water in modified waterways and harbors. This is most often achieved through structural engineering. In contrast, deficiencies in ecosystem integrity and health can be approached globally by restoring and sustaining the viability of all parts and processes contributing to natural integrity. This is most often achieved by removing the effects of past structural engineering.

Human demands for cultural development assures that most ecosystem restoration efforts will seek no more than partial restoration of the entire ecosystem, resulting in some mix of natural and human-modified properties. Even so, partial restoration is expected to lead to self-regulating functions and sustainable ecosystem states. The concepts of cultural integrity and ecosystem health are the basis of this expectation (which go back nearly a century in natural resource management). Expectations are complicated, however, by the growing realization that functions and structures respond to restoration at different rates.

**Restoration Responses of Function and Structure**

Ecosystem-level functions —such as production and biomass accumulation, materials cycling, and ecosystem resilience— are dominated by the performance of a few common species. The rare species provide some functional refinement and a lot of functional redundancy, which contributes to resilience following exceptionally intense disturbance. Partial restoration that recovers the common biodiversity typically captures a large fraction of the natural functions supporting many services may fail to restore much of the scare biodiversity expressing the unique genetic information held in ecosystems.

Much of the important function and service associated with maintaining unique genetic information is linked with globally scarce species. In addition to potential resource-development value, those species provide functional "backup" that replaces common species when ecosystems undergo exceptional stress. Scarce species are not missed in most ecosystem functions under ordinary conditions, but are significant for sustaining natural ecosystem resilience and management options well into the future. The scarcest resources globally (e.g., species vulnerable to extinction) are among the most significant of those resources, and the most challenging to restore. Recovery of scarce species involves much greater uncertainty and risk than the restoration of common species and associated functions. This risk is often a reason given to avoid targeting scarce species, especially in small restoration projects, and instead emphasizing restoration of more common function and structure. (That rational of course misses the restoration point entirely). A fundamental way to control such risk is to scale up the recovery of ecosystem resources to a more inclusive level of influential landscape and community composition. Of course that is more expensive.

**Ecosystem Indicators of Resource Significance**

Ecological concepts pertaining to naturalness, such as ecosystem integrity and natural biodiversity, are service-neutral constructs (where services are defined as things that ecosystems do which contribute to satisfying human wants and needs). Therefore, ecological indicators of naturalness will not necessarily indicate or track the types and levels of ecosystem services that may result from change to a more natural state. Concepts of ecosystem integrity and biodiversity gain service- and resource-significance meaning only after variations in ecosystem integrity and biodiversity are calibrated against variations in recognized service output from the same restoration action. The non-monetary metrics chosen for project evaluation should indicate the desired change in the resources. For example, the ecosystem service that sustains a rare species might better be measured as number of recruited adults than total number. Once resources of significance are recast as ecological indicators providing the desired service, models can be selected or built to forecast indicator-output response to with- and without-plan conditions. When the indicators of significance are not greater naturalness alone, these models will need to accommodate more than one measure of ecological output.

Regardless of whether greater naturalness alone is a significant resource response, Corps policy ordinarily expects a more natural (or naturalistic) condition to result. Indicators of naturalness and indicators of more specific resource significance can respond quite differently to restoration plans that propose only partial ecosystem restoration. Commonly used biodiversity indicators or other indicators of greater naturalness often are composed of the more common parts and processes in ecosystems while the parts and processes of greatest significance often are among the scarcest resources. The increment of naturalness indicated by a project plan may not be sufficient to support the increments of resource output and services that justify the restoration investment. These differences converge into one quality indicator as restoration approaches a fully restored ecosystem. If resources of significance are confidently associated with full restoration of ecosystem naturalness, any good indicator of full restoration will also confidently predict recovery of the resources of significance (but only with respect to completeness of knowledge about the natural reference conditions). When the proposed restoration is only partial, planning models and methods (and the evaluation framework more generally) need to account for a possible differential response of indicators of naturalness and indicators of significant resources in the range of humanly disturbed ecosystems over which the partial restoration is to occur.

**A Widely Applicable Indicator of Environmental Value**

No widely applicable non-monetary measure of change in environmental value exists, but there may be an inclusive measure of NER contribution, depending on how Corps policy is interpreted. Energy units, genes, or other universal measures of ecological product are too general to capture the different values perceived in their diverse expressions. Short of a full measure, and in keeping with the evaluation criteria for NER, the common biodiversity indicating the natural integrity of ecosystems is one possibility if the primary objective of restoration is simply greater naturalness of resources responding to habitat improvement. However, different values are likely to be placed on specific expressions of biodiversity by at least some stakeholders in the restoration result. Even if that were not the case, natural integrity does not seem to hold up to the need for a national-level of "standard-unit" measure. Whereas the biodiversity existing locally in an ecosystem can be gauged against fully natural sites within an ecosystem, no way to compare across ecosystems is evident. Two ecosystems of equal integrity can have very different biodiversities. It is also difficult to determine what increments of biodiversity mean in terms of their relative naturalness.

Corps policy indicates that the NER contribution is to be made up of ecological resources associated with "terrestrial and aquatic species", which includes numerous resources used for commodities, recreational, and aesthetic services, all of which are excluded by policy. A promising measure of NER contribution is based on the genetic uniqueness of scarce biodiversity in the form of species at risk of global extinction. The greater the uniqueness and vulnerability of species at risk, the greater is the deficiency in service and value based on future resource development and management options at risk of loss. Option value is not situational, and its indication in a uniqueness-vulnerability index can be made comparable within and across ecosystems at scales varying from local through national (or international) levels.

A uniqueness-vulnerability index (a natural currency) is presently conceivable (if crudely so to start) and is consistent with institutional, public, and technical indicators of significance, such as the Endangered Species Act and conservation status designations by the state natural heritage programs. Existing indicators of uniqueness/scarcity include various national and international "red lists" of species vulnerable to extinction. A leading source of such information is the database, NatureServe, that is maintained for the state natural heritage programs in the United States.

This uniqueness-vulnerability index falls far short of representing all value. This index is based on the relative scarcity of species traits and genetic information, and it places high value on maintenance of the scarcest species for future management options, including restoration options. It places very little value on common species, despite the many ecological services that are provided by common species (They dominate the production, biomass and other ecological process underlying many services). Most utilitarian values, including NED, are associated with relatively common species, such as the species that support hunting, fishing and much other outdoor recreation. Yet, there may also be services and values as yet to be revealed that fall outside this index and the NED monetary index to value. Until those values are revealed, however, a uniqueness-vulnerability index is a good interim measure of NER contribution worthy of serious consideration.

## Model Use and Development

**Section 4** describes and examines existing types of ecological models with respect to their applicability in ecosystem restoration planning.

### Index Models

Corps environmental planning history has been much more closely tied to index models than to actual-output estimation models, despite a long history of developing actual output estimates for hydrologic analysis. The P&G provides only one substantial example of an ecosystem evaluation procedure, which is a single-species index model designed for impact mitigation analysis required by the National Environmental Policy Act. Many such species-habitat indexes were generated in the early 1980s, just in time for the new environmental improvement authority in 1986. Thus species-based index models quickly became the model of first resort. Other probable reasons for choosing them over actual estimation techniques include a deficiency of long-term ecological databases (unlike hydrologic databases), substantial Corps professional uncertainty in ecological theory and application, and a much more limited computing environment than now exists.

The most fundamental difficulty with species-habitat index models is that they typically are not inclusive enough of all community-habitat interactions. They are not indexed with respect to a fully natural ecosystem condition or to any other indication of naturalness, but instead are indexed with respect to variation from an optimum condition for the chosen species. Thus, the meaning of the index with respect to naturalness can be confusing and may lead to the development of plans that focus on artificial enhancement (rather than restoration) of habitats when the optimum habitat condition is targeted.

More recently, a number of community-level index models have been developed, at least in prototype, and offer a number of improvements over species-level models, because they are indexed to the most natural condition, rather than to some optimum habitat condition. Models used in the Index of Biotic Integrity, Wildlife Community Habitat Evaluation, and Riverine Community Habitat Assessment and Restoration Concept, for example, anchor their maximum index value to native species diversity or other native biodiversity measure existing in the most natural habitat state determined from existing reference conditions. This is also true for the ecosystem functional capacity index models of the Hydrogeomorphic approach, which are anchored in the most natural ecosystem state indicated by reference sites.

These direct measures of ecosystem naturalness can be useful for restoration planning to the extent that a more natural or naturalistic condition is integral to project objective achievement. However, unless the entire community is an ecological resource of national significance, or there is a known direct relationship between the community index and the condition of the significant resource, some other measure(s) of resource significance is needed to determine if the forecasted gain in naturalness is socially significant. For example, the habitat conditions input to a community or ecosystem model that achieve some increment of greater naturalness could be used for inputs to a second set of models that forecast the condition of specific significant resources (e.g., rare species, recreational species, change in water discharge). In this way, the second set of models would be used to evaluate the significance of the effect of restoring a more natural condition. Best use of models in this way demands a lot of a good concept model of the target ecosystem and choice of habitat inputs. Few existing models fully qualify for this level of use, but existing models can be adapted if carefully interpreted in the context of a good concept model.

A common deficiency of all existing index models, regardless of their indication of relative naturalness, is their focus on a local planning perspective. They tend to ignore large-scale landscape features and processes that can be very important for restoring and sustaining significant resources. They also tend to ignore the need to evaluate contributions of significant resources through a national perspective as well as the local perspective. The linkage to a national perspective is left for other methods (e.g., best professional judgment). These deficiencies could be addressed in development of future index models, but would be faced by substantial conceptual difficulty arising out of the interactions that occur between the size and patters of habitats in the modeled area and the quality of each unit of habitat in the area.

**Actual Estimation Models**

Actual estimation models include physical, statistical, and process models. Actual estimates of outputs have an obvious advantage for communicating expectations explicitly rather than implicitly based on knowledge of indexed reference conditions. Physical and statistical models have limited use in relatively simple restoration situations where the desired conditions and measures are quite obviously indicated in a basically intact natural setting with very little human impact. Unlike process models, they are not very flexible or portable.

Most process models also focus on local perspectives (e.g., lakes, wetlands, stream segments), but nevertheless provide many advantages, including the potential for incorporating complex interactions between size and patterns of the area models and interactions with quality of individual units of habitat. The most useful process models simulate dynamics in time and space. Their greatest shortcoming is their need for process understanding, which often is time consuming and expensive to obtain. However, they are unsurpassed for situations demanding explicit integration of non-linear relationships leading to multiple outputs and time dependent feedbacks. They are the most potentially useful when it is impossible to capture all benefits in a single ecological output indicator and multiple indicators are needed. Recent model advances enable simulations of spatial interactions at large scales.

Although process simulation models have no inherently better predictive attributes than other models, and less so than statistical models, their workings and outputs are more explicit, often making them superior communication tools for tradeoff analysis in controversial planning environments and for organizing a rigorous adaptive management framework. They are unsurpassed for organizing new information adaptively into the model structure as lessons are learned. Process models such as CASM, a lake ecosystem model, show a strong potential for developing generic ecosystem models (requiring local calibration). Other process-based models/methods, such as ATLSS, an Everglades restoration model, show the potential for representing spatial dynamics in outputs. Process models also show the greatest potential for comprehensive forecasting of those physical outputs underlying all economic and environmental consequences of plan implementation in multipurpose studies.

### Picking The Best Model

Determining the best models for any restoration context depends on the complexity of ecosystem alteration that has occurred, the complexity of objectives to be achieved, and the risks that would be incurred if the ecosystem outputs should fail to become established as forecast. Just about any rigorously applied model type will suffice for situations where there has been very little ecosystem change from the natural state, the condition to be restored is closely connected to the restoration site, full restoration is targeted, the source of the deficiency in resources of significance is easily identified and removed, and the decisions are not controversial. As ecosystem and planning conditions grow more complicated, however, the advantages of process simulation models begin to outweigh the relative accessibility and cost advantages of index models.

## Policy Standards for Plan Evaluation, Comparison and Selection

**Section 5** reviews the Corps planning framework for ecosystem restoration by way of outlining the analytical framework used for traditional civil works missions, and how it has been adapted for restoration planning.

### Economic Development Projects

Corps planning regulations for traditional civil works purposes define the federal objective in project planning as contributions to "National Economic Development' (NED) as represented by increases in the net economic value of goods and services. That is, traditional purposes are pursued based on a utilitarian philosophy that recognizes contributions to the satisfaction of human preferences as the ultimate goal of civil works activities. Following that standard, project plans must be evaluated in terms of their monetary (NED) costs and benefits, and used within a benefit-cost analytical framework to compare project plans in terms of net NED benefits (monetary benefits less costs). The recommended plan for federal action is to be the alternative plan that provides the greatest positive net benefits that is also consistent with environmental protection.

**Ecosystem Restoration Projects**

The plan evaluation, comparison and selection standards established by Corps regulations for the NER purpose different in several important ways from those applied to NED purposes. First, Corps regulations stipulate that restoration outputs must be measured in physical or biological units of some kind that reflect resource "value". Second, since NER outputs are to be measured in metrics that are incommensurable with money measures of project costs, benefit-cost analysis cannot be used to evaluate and select plans in terms of a net benefits criterion. Instead, Corps regulations require the use of "cost-effectiveness analysis" for plan comparison, in which the monetary costs of each alternative plan are weighed against the non-monetary level of NER output produced by the plan. Cost-effectiveness analysis provides a way to identify the set of cost-effective plans representing the least-cost means of producing different achievable levels of NER output. Further, Corps regulations specify that "incremental cost analysis" should be performed to identify the marginal cost per unit of output gained from moving from one cost-effective plan to the next higher-output, cost-effective plan. That analysis can help planners to identify plans for which the added NER output achievable may not justify the additional cost required to achieve it.

Together, cost-effectiveness and incremental cost analyses (CE/ICA) serve to narrow and illustrate tradeoffs among the set of NER plans considered for selection. But since plan costs and benefits are expressed in incommensurable terms, these analytical procedures cannot identify a "best" plan in an objective way comparable to the positive net benefits criterion used for the selection of NED project plans. Instead, Corps regulations say that restoration project plans can be selected based on a subjective determination that non-monetary outputs are worth monetary costs, provided that the selected plan is cost-effective and NER outputs are shown to be "significant" based on institutional, public, or technical recognition of importance.

**Multipurpose NED/NER Projects**

For multipurpose NED/NER projects, Corps policy says that plan selection shall attempt to maximize the difference between project benefits--both NED (monetary) and NER (non-monetary)—and project costs, and strike the best balance between the two objectives. As in the single purpose NER context, this justification standard necessarily requires a subjective determination of the 'best" plan since NER outputs and NED effects (benefits and costs) are evaluated and expressed in different metrics.

Planning guidance suggests that the evaluation and comparison of NED/NER plans should rely on a combination of benefit-cost analysis for the NED purpose, and CE/ICA for the NER purpose. However, that would not possible when plans are characterized by joint costs – or costs that simultaneously produce both NED and NER outputs. Joint costs should be common for NED/NER projects since the primary rationale for pursuing a multipurpose project instead of separate single purpose projects is potential efficiencies that can be realized by exploiting opportunities to jointly produce desired outputs. If a dollar's worth of plan costs serves both NED and NER outputs, these costs and benefits must be considered together for plan comparison. This can be readily accomplished since plan costs and NED benefits are both measured in dollars and are recognized by Corps regulations as fungible (i.e., a dollar's worth of benefit for a formulated NED purpose exactly offsets a dollar's worth of plan cost). Given this, the CE/ICA framework can be implemented using a measure of *net* plan costs calculated by subtracting NED benefits yielded by the plan from the financial costs of implementing the plan.

A slightly different approach for analyzing the economic efficiency implications of NED/NER plans would extend the two-criteria cost-effectiveness framework to one defined over multiple criteria. So, for example, in a planning case in which plans have been evaluated in terms NED benefits, implementation costs, and a single measure of NER output, an efficiency analysis based on these three criteria would identify the set of plans for which more NER output could not be obtained through choice of another plan without realizing higher plan costs or lower NED benefits. This framework is also appropriate in the single or multipurpose planning case in which NER outputs are evaluated in multiple, incommensurable metrics.

## Potential for Monetary Evaluation of Restoration Outputs

**Section 6** reviews and assesses the potential use of a monetary standard for evaluating restoration plans, including the possibility that the benefit-cost analytical framework used to compare and recommend traditional civil works project plans could be applied to the ecosystem restoration context.

The concept of economic value rests on the presumption that the welfare of any individual derives from the satisfaction of that individual's preferences. Acceptance of that premise implies that the tradeoffs that a person makes as he or she chooses less of one good in favor or more of another good reveals something about the economic value of this tradeoff to the individual. Formally, the economic value of some change (tradeoff) to an affected individual is the amount of monetary compensation (positive or negative) that the individual would need in order to maintain the same level of preference satisfaction with the change as without the change. The conceptually valid measures of economic value for some policy change are "willingness to pay" (WTP) compensation for policy benefits, and "willingness to accept" (WTA) compensation for policy costs. Total benefits are defined as the sum of WTP estimates for each individual who stands to gain from the new policy, and total costs are defined as the sum of WTA for each individual who stands to lose from the new policy.

### Measurement of Economic Value

Use of a benefit-cost analytical framework for evaluating ecosystem restoration plan alternatives would require an accounting of the total economic value associated with the expected change in all of the service flows resulting from those plans. Market prices provide the basis for estimating economic values associated with changes in the flow of goods and services that are traded in competitive markets. But since ecosystem services generally are not traded in markets, there exist no market prices that can be exploited to estimate the economic value of changes in service flows. To address this limitation, economic methods have been developed to estimate "shadow values" for non-market services that, in theory, represent the market prices for services that would emerge if the services were traded in competitive markets.

**NED Evaluation of Traditional Outputs**

The Corps has long faced the need for non-market valuation since traditional civil works outputs generally are not traded in competitive markets. However, many traditional outputs have close market counterparts that facilitate benefits estimation based on *change in net income*. So, for example, the benefits from enhancing waterway transportation links are assessed as cost savings to commercial navigation shippers, and the benefits from enhancing flood regulation services are valued in terms of property damages avoided. Similarly, the benefits of new sources of water supply and hydropower can be estimated based on the *cost of the most likely alternative.*

In general, the valuation of traditional civil works outputs such as commercial navigation, flood control, hydropower and water supply has been readily possible for two main reasons. First, changes in the underlying ecosystem service flows (waterway transportation links, flood storage and diversion capacity) resulting from management action are intensive and largely involve physical relationships that are well understood and predictable. Thus, for many traditional outputs the types of non-economic information needed for valuation is readily obtained. Second, as outlined above, these outputs generally have close market counterparts that provide market evidence of economic value.

**NED Evaluation of Restoration Outputs**

In general, the economic techniques outlined by Corps guidance for valuing traditional civil works outputs are also applicable to estimating values for the types of natural service outputs likely to be associated with ecosystem restoration. This does not mean that valuation prospects for restoration outputs are generally favorable, however. Indeed, the information needed for valuation in this context is often not readily estimable. Restoration effects on natural ecosystem services are often subtle and involve complex biological relationships that are not well understood and predictable. These factors impede development of the non-economic information needed for valuation. And natural services often affect human welfare in ways that have no close connection to the use of market goods. In the extreme, people may hold “passive use values” associated with the knowledge of the existence of certain ecosystem services (e.g., sustenance of endangered species) independent of current or anticipated future use of these services. These factors severely limit the extent to which market data can be exploited to infer values for restoration outputs.

In addition to these technical hurdles, some economists and other professionals argue that the utilitarian concept of economic value does not tell us everything we need to know about the desirability of environmental protection and restoration activities. Challenges from these critics could be expected to hinder the political acceptability of adopting a framework for restoration planning that turns on the economic valuation of restoration outputs.

Nevertheless, in some cases it should be technically possible to estimate economic values for certain restoration effects that could be used to inform decision making in ways that are politically acceptable. An obvious example is when restoration project plans measurably affect traditional NED outputs such as flood regulation. In such cases, these effects should be valued and used within the CE/ICA framework for evaluating and comparing alternative project plans in terms of net monetary effects and non-monetary, ecological effects.

## A Strategy for Improving Environmental Benefits Analysis

**Section 7** presents conclusions and a draft strategy for improving environmental benefits analysis, including immediate, intermediate and long-range components that address models and methods, capabilities in using models, and policy and guidance.

### Conclusions

The study discovered no universal unit for expressing ecosystem restoration benefits in non-monetary terms that can adequately evaluate the full range of restoration plan effects. However, the notion of “biodiversity associated with scarce species” (as defined by uniqueness and vulnerability) could be pursued to develop a standard non-monetary measure of resource significance that could help discriminate among NER investment choices.

This notion of “scarce species biodiversity” can be distinguished from the more comprehensive concept of biodiversity in that it focuses on that subset of species, communities, guilds and ecosystems designated to be of *significance* because the loss of their unique traits to extinction would be irreplaceable. This significance is technically identified in numerous scientific reports, including the work of the WWF, TNC, and state natural heritage programs. It is also indicated institutionally by the Endangered Species Act, which establishes a national objective to recover threatened and endangered species and necessary habitat support to an unlisted status. Pursuing this measure would be compatible with the habitat-based emphasis of the current Corps policy, and with the policy emphasis on resource scarcity as an indicator of significance. The standard units (see discussion in Section 3) would be based on characteristics of vulnerability and uniqueness, using methods developed by conservation biologists, and taking into account global rather than only localized significance. For example, while some significance may be inferred by plans supporting the North American Waterfowl Management Plan, greater significance would be attributed to plans that support a species such as black ducks, which are rare compared to mallards, which are included in the plan but are neither rare nor vulnerable.

Such “scarce species biodiversity” may not be the only relevant measure of resource significance that contributes to NER output, but placing emphasis and priority on such outputs can be supported because the recovery and protection of scarce resources determines the limits of future management options, including restoration options.

The study also found that a variety of ecological models are useful in formulating and justifying ecosystem restoration investments based on forecasts of ecosystem-level conditions (with more or less human effect) and specific outputs of significant resources. The models can be usefully applied alone or in combination, depending on the circumstances. In the near term, a combination of community-habitat index models that forecast naturalness (including those such as IBI), and species-habitat index models that forecast suitability of the more natural state for specific resources of significance can provide a sound basis for evaluating plan effects. In those instances where *the more natural condition in itself* is identified as the resource of social significance, ecosystem-level biodiversity models and methods that are habitat based (e.g., IBI, WCHE, HGM) may serve satisfactorily, once calibrated. Model selection depends on the extent to which a more natural condition or more resources of significance should guide restoration formulation and evaluation. Corps guidance is not as clear as it could be regarding desired outcomes of degrees of natural conditions versus resources of significance.

This conclusion does not, however, address the limitation that habitat-based indicators of NER benefit are unlikely to capture all of the Federal interest affected by restoration plans (as noted by the NRC). In addition, relatively few species-habitat models have been specifically developed for rare resources. Other models, such as functional capacity indices (e.g. HGM, water storage, organic export) and process simulation models (CASM) are applicable for the multi-output analysis of benefits that appears to be required for multipurpose planning. Ecosystem process models have the advantage of generating more theoretically defensible and explicit results unsurpassed for communication and adaptive management, but are more costly. All existing models have shortcomings requiring substantial development effort, but especially so for the process simulation models.

The study also concludes that significant technical obstacles preclude the economic valuation of all possible restoration outcomes that could potentially be evaluated in monetary terms. Furthermore, whether or not the utilitarian concept of economic value is the appropriate standard of "value" for evaluating restoration outcomes is open to question. Economic value may not indicate everything that stakeholders need to know about the desirability of restoration projects. This suggests that the current policy guidance that recognizes non-monetary NER outcomes as a category of effects separate from monetary effects is appropriate for evaluating restoration projects. However, a greater level of policy clarity is probably needed to help planners determine the appropriate restoration objectives and valuation standards for restoration planning.

Use of evaluation criteria that includes both non-monetary and monetary effects does not reduce the need for efficiency analysis in the NER planning context, and this need is recognized by Corps guidance. The cost-effectiveness analytical framework for single-purpose NER planning is very useful for evaluating the opportunity costs and marginal tradeoffs among alternative plans. That framework, which is essentially equivalent to the old P&S efficiency framework that plotted net NED effects against some measure of environmental quality change, is also applicable to multipurpose NED/NER planning, and can be readily extended to a multiple criteria efficiency analysis when NER outputs are best expressed in multiple, non-commensurate metrics. The cost effectiveness framework is less discriminating as the number of choice criteria increases, making identification of more inclusive metrics an important pursuit. A focus for improving ecosystem restoration benefits analysis in the near term is to identify the monetary and non-monetary indicators of output needed to capture all significant effects, and ultimately to reduce them down to the minimum achievable.

**Strategy**

Preliminary ideas for a multi-component, three-stage strategy for improving environment benefits analysis are offered for consideration. The proposed strategy involves overlapping (I) *near-term, (II) intermediate,* and *(III) long-term* components. Within each component is attention to model development, staff model application capabilities, and policy and guidance issues. All three components would start as soon as possible, but would begin to produce useful results on different time horizons. The extent and timeliness of result will depend on initiation dates, investment levels, and concentration of effort. A number of ongoing efforts are identified that will or could contribute to this strategy in the Section 7 strategy discussion.

The *near-term* or **Incremental** stage (2 to 3 year horizon), addresses the requirements of the current Corps planning regulations, seeks modest advances in improving environmental models, and emphasizes improving staff selection, adaptation and application of a broader set of existing ecological assessment models. Broadening staff proficiency should enable more informed application of new models as they are developed, in addition to improving environmental benefits analysis now.

- Modest model improvements would be made by moving from reliance on single-species index models alone, to greater use of community-based index models, either alone or in combination with single-species index models. Ongoing efforts within the EMRRP program, such as the development of templates for community-index models, should contribute to such improvements in the near term.

- A protocol is being developed for selecting ecological assessment models and methods for use in ecosystem restoration planning. It will be published as a reference guide that summarizes different model types, attributes, limitations, and utility in the 6-step, Corps planning process.

- Concepts of and approaches to environmental benefits analysis need to be incorporated into a number of courses, workshops and other forums, at appropriate levels of detail, including the new Planner Core Curriculum, as well as over a dozen PROSPECT courses. Collaboration among IWR staff and Corps course instructors will help assure consistency and comprehensiveness in course instruction material refinement and presentation.

- A web-based tool catalog is being developed as part of the SMART R&D program. Both this effort and the model selection reference are likely to be linked within the web-based EMRIS system, assuming sustained funding support for the efforts.

- Workshops to apply the model-selection reference document to actual studies in a district would help improve district staff capabilities, inform invited staff from other agencies, and refine the instruction material for use in future courses and workshops.

- Future policy and guidance refinement should explicitly consider the issues raised by the discussions of ecological and methodological concepts presented in this report. They provide a theoretical basis, and identify some unresolved issues for informing improvement of NER evaluation.

- The NER concept is being examined in a FY03 policy study as a federal objective and basis for formulating ecosystem restoration projects. The NER study will examine the potential usefulness of the concept of ecosystem services for defining NER as a formulation construct and for developing a set of standard methods and metrics for characterizing and evaluating NER outputs

The *intermediate,* or **Next Generation** stage (5-year horizon), would pursue a fundamental rethinking of the NER objective and desired outputs. It would more intensely pursue the idea that ecosystems provide significant mixes of ecological services, the benefits from which can be compared with traditional NED outputs, and the possible advantages and practicality of defining an NER account that specifies these services. Further, it would seek to improve the ability to simultaneously evaluate multiple outputs indicating resources and services though the use of ecosystem process simulation models at proper landscape scales. New analytical frameworks for multipurpose NED/NER planning would be explored, including the opportunity-cost framework recommended by the *Principles and Standards* several decades ago for evaluating tradeoffs between economic and non-monetary environmental effects.

- Efforts to develop and refine ecosystem process models to forecast resource responses and associated outputs to restoration alternatives should closely consider the evaluation frameworks used in Corps planning, and inform the further evolution of those frameworks.

- Future efforts should investigate the development of a metric based on the biodiversity of scarce species, and its usefulness in determining the significance of forecasted NER plan contributions to significant resources.

- Research programs such as the EMRRP, SMART and TOWNS, and the newly formed Environmental Modeling and System-wide Assessment Center (EMSAC), along with the EMRIS system, could play key roles expanding the applications of existing models and developing new models. These programs need to be refined to more effectively address issues pertaining to environmental benefits analysis identified and examined in this study.

- Proposed work within the EMRRP ('04) would develop a framework that links habitat analysis, dynamic process modeling, and spatial statistics for application in aquatic systems.

- Potential applications of the Ecosystem Functions Model (EFM) beyond the Sacramento-San Joaquin basin, as well as the Watershed Analysis Tool (WAT) being developed as part of the Flood and Coastal Systems R&D initiative should be explored for potential advancements in ecosystem restoration planning.

- The formation of *model application assistance teams* could facilitate improving model selection and use of model output information in investment and management decision making.

- Policy and guidance efforts during this stage would further refine the NER concept and outputs, relative to ecosystem goods and services, along with alterative analytical frameworks useful in Corps planning, especially for joint NED/NER projects. Emphasis would be placed on better integration of project development, including ecosystem restoration, in its landscape context, to better serve the NED/NER Federal objective.

- The potential applicability of the concept of *ecosystem services* in water resources planning would be further explored for its usefulness in differentiating NED and NER and assuring that all of the monetary and non-monetary costs and benefits are considered.

- Procedures should consider the "sustainability" philosophy expressed in the PCSD (1996), and evolving through implementation of the Environmental Operating Principles.

- A broader notion of environmental analysis should be considered, which integrates the "NEPA process" into the P&G/P&S planning process, eliminating differing standards and principles for evaluation for ecosystem restoration planning and environmental impact assessment.

Over a *longer-term* (beyond a 10-year horizon), efforts could be made to explore more comprehensive economic valuation of ecosystem services. This **Monetization** stage would attempt to link ecological process simulation with economic valuation methods for the evaluation of restoration outcomes in economic terms. If deemed practical and acceptable, this might ultimately lead to the development of standard analytical tools that would allow more NER outcomes to be evaluated in NED terms, thereby reducing the total number of non-commensurate choice criteria that would need to be considered for efficiency analysis.

- Technical obstacles to comprehensive monetary accounting of restoration benefits need careful consideration, including the uncertainties associated with forecasting ecological outputs from alternative plans and the limitations in methods for measuring non-market benefits of service outcomes that affect the quality of human life in ways that have no close connection to the use of marketed goods.

- Also to be considered are challenges from critics who question the acceptability of economic valuation for environmental benefits, which could hinder the political acceptability of adopting a monetary standard for evaluating and justifying restoration projects.

- To the extent possible, the Corps should pursue the environmental benefits analysis improvement strategy in conjunction with other Federal and state agencies that can contribute to and benefit from these efforts.

- Ongoing work within the Decision Methodologies Research Program will contribute to this pursuit. These include identifying recent and ongoing district studies that monetized environmental outputs, identifying applications in other agencies, and a literature review. Additionally, a test case has been proposed that would apply monetization to a completed ecosystem restoration project, to examine whether and how this information could have been useful in decision making. The potential analysis of air quality benefits, from reduced emissions attributed to inland waterway shipping relative to land-based transportation modes is also being examined.

# Section 1. Introduction

## 1.1 Background

The U.S. Army Corps of Engineers (Corps) now pursues "National Ecosystem Restoration" as a priority Civil Works purpose on par with traditional "National Economic Development" purposes such as commercial navigation and flood damage reduction. Further, the Corps' new Environmental Operating Principles say that Civil Works planning should strive to achieve "environmental sustainability" and "seek balance and synergy among human development activities and natural systems by designing economic and environmental solutions that support and reinforce one another."[1]

In traditional Civil Works planning, the stated Federal objective is to contribute to National Economic Development (NED) consistent with environmental protection. Following that overarching goal, desired economic outputs (e.g., commercial navigation) are evaluated in monetary terms, alternative plans are compared using benefit-cost analysis, and plan selection is based on a national economic efficiency standard (positive net benefits criterion). A somewhat different framework has been established for ecosystem restoration planning, however.

According to Corps planning regulations, the Federal objective in ecosystem restoration is to contribute to National Ecosystem Restoration (NER), where contributions are defined as "increases in the net quantity and/or quality of desired resources ecosystem resources", and "measurement of NER is based on changes in ecological resource quality as a function of improvement in habitat quality and/or quantity expressed quantitatively in physical units or indexes (but not monetary units)." [ER 1105-2-100; Section 2.2 b]. Since restoration outputs are to be characterized and evaluated in non-monetary terms, traditional benefit-cost analysis and plan selection based on a net benefits criterion are not applicable to ecosystem restoration planning. Instead, Corps planning guidance says that ecosystem restoration plans are to be compared using cost-effectiveness analysis to ensure that the least-cost plan is identified for any achievable level of non-monetary restoration output. A cost-effective plan can then be recommended based on a subjective determination that non-monetary outputs are worth the costs of producing them, in consideration of the "significance" of project outputs as indicated by institutional, public or technical recognition of importance. Corps planning guidance emphasizes the importance of the significance concept for helping planners to, firstly, determine the Federal interest in restoration planning for some area, and secondly, judge whether the improvement in resource output associated with some project plan warrants its cost.

The recent emphasis on assuring returns from Federal investments, along with the reality of budgetary constraints, have resulted in a renewed interest in the methods used to evaluate the outputs of environmental projects and programs. Importantly for the Corps, the adequacy of the methods currently being used within the Civil Works program for characterizing and evaluating the environmental outputs of ecosystem restoration projects has been questioned in a number of forums. For example, the National Research Council (NRC) report *New Directions in Water*

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[1] *U.S. Army Corps of Engineers Environmental Operating Principles and Implementation Guidance*. March 26, 2002.

*Resources Planning for the US Army Corps of Engineers* suggests that the Corps needs to move beyond its current reliance on habitat assessment methods, such as the “Habitat Evaluation Procedure” (HEP), for evaluating the restoration projects. On this point the NRC report says, “The difficulty with HEP and similar methods is that they capture only part of the national interest” in ecosystem restoration (NRC, 1999a; p. 77).

### 1.2 Purpose

This report was motivated by a recognized need to improve the characterization and evaluation of the environmental outcomes of Corps projects. Toward that desired end, the report identifies and examines a diverse set of policy and technical issues related to improving environmental benefits analysis for Civil Works planning. As used here, the term “environmental benefits analysis” refers to the development of an evaluation philosophy, framework and complementary analytical tools to aid Corps project evaluation and selection, but is not intended to imply a planning framework for ecosystem restoration that involves assigning dollar values to restoration outputs (though the prerequisites for economic valuation are discussed). Rather, this report focuses on the analysis and science related to assessing the anticipated results of ecosystem restoration as expressed in non-monetary metrics. The development of this report was informed by several workshops conducted to engage various experts from within and outside the Corps in the search for practical approaches to environmental benefits analysis consistent with traditional water resources planning and evaluation principles. Development of the strategy was informed by field responses to a survey conducted in January 2003.

Environmental benefits analysis is applicable to ecosystem restoration projects, the broader ensemble of environmental enhancement and mitigation features, as well as water resources projects with mixed economic and environmental outputs. As such, the ideas and methods offered are consistent with economic-ecologic linkages and multiple objective tradeoffs that are inherent in the guiding principles and philosophy of the “Principles and Guidelines” (P&G) [Water Resources Council, 1983], the earlier “Principles and Standards” (P&S) [Water Resources Council, 1973], and the Corps’ own planning guidance (ER1105-2-100) which is a further procedural embellishment of the P&G, with updated policies and procedures.

The primary focus of this report relates to the identification and assessment of alternative metrics and analytical procedures for characterizing and evaluating environmental outputs in non-monetary terms. A secondary focus relates to the identification and assessment of alternative plan comparison frameworks for projects plans formulated at least in part to serve ecosystem restoration. With regards to these issues, the report addresses the following questions:

- What non-monetary metrics of environmental quality change may have wide applicability for characterizing and evaluating ecosystem restoration outputs?

- What plan comparison frameworks and procedures are available for plans formulated to serve ecosystem restoration as well as mixed economic and environmental objectives, and what are their strengths and weaknesses for illuminating the economic efficiency implications and tradeoffs among plans?

The report concludes with a proposed strategy for improving environmental benefits analysis that includes several specific options that reflect the current practices, limits and constraints on conventional analysis and rely on ideas, practices and technologies that are currently available, but not yet widely implemented. The presented options are intended to crystallize a few possibilities for improving environmental benefits analysis without suggesting that other possibilities are infeasible. Ideas are presented that may contribute to shaping the next generation of evaluation principles and analytical tools for environmental benefits analysis, both within the Corps and, hopefully, among other Federal agencies confronted with comparable responsibilities

### 1.3 Organization of Report

The report is organization as follows. The remainder of Section 1 overviews the broad need for improved environmental benefits analysis within and across Corps programs and those of other Federal agencies. Section 2 reviews the planning framework for ecosystem restoration as defined by Corps planning regulations and supporting policy information, introduces the concept of ecosystem services, and discusses the relevance of the service concept for ecosystem restoration planning. Section 3 presents a critical review of ecosystem concepts that might be useful for characterizing and evaluating ecosystem restoration outputs, including concepts of resource "naturalness" and "significance". It also reviews contemporary ecological theory since an understanding of a number of ecosystem principles is critical for establishing meaningful restoration objectives and formulating and evaluating restoration alternatives. Section 4 provides an overview of the types and attributes of ecological models and methods used by the Corps and other agencies in an attempt to assess the current state of applications and innovation in contemporary project planning and evaluation. Section 5 provides an overview of the planning framework used by the Corps for traditional NED purposes, and how it has been adapted for the ecosystem restoration purpose. It also reviews alternative plan comparison frameworks for project plans formulated to serve both environmental and economic objectives, and comments on their strengths and limits for illuminating the economic efficiency implications and tradeoffs among plans. Section 6 explores important technical and conceptual issues relating to the possibility for using a monetary plan evaluation and selection standard for ecosystem restoration projects. Finally, Section 7 suggests a broad strategy for developing improved benefits analysis models and approaches that includes multiple options.

## 1.4 Why is Improving Environmental Benefits Analysis Important?

The past 15 years has seen the introduction of many new ideas on, and programs for ecosystem management, restoration, remediation and mitigation, which, when added to many other overarching concepts of sustainable development, cumulative impacts, biodiversity, endangered species protection and “global change”, have created a conceptual and philosophical morass of confusing and ill-defined aims. Understandably, the Corps and most other Federal resource management agencies (e.g., USFS, NRCS, BOR, NMFS, EPA, FWS, BLM) have found it difficult to develop evaluation procedures and complementary analytical tools that translate the principles of ecosystem management into accepted conventional practice. The problems are exacerbated as each agency develops its own version of procedures that reflect their respective missions and traditions of analysis. Problems typically come to the fore in situations where multiple agencies are involved in sharing resource management responsibilities in a given area.

Inherent in improving environmental benefits analysis is not only the notion of improving evaluation of ecosystem responses and associated outputs related to management measures, but also the need to improve analysis of tradeoffs among environmental, economic and social objectives and effects. Changes in National water resources priorities, public demands and the Corps’ missions have fostered a need for improved environmental analysis. However, these factors are not the only catalysts for pursuing improvements in environmental benefits assessment. The recommendations of the recent National Research Council Report (1999a) on the Corps planning procedures, call on the Corps to:

- Thoroughly review the P&G, and modify to incorporate contemporary techniques and ` public values,
- Improve analytical techniques for environmental benefits and detriments assessment, and;
- Develop a standardized set of tools for quantifying environmental benefits and costs.

Addressing these contemporary demands on analytical capabilities and evaluation frameworks can foster the principles of sustainability and draw upon a wealth of literature and theory. Three decades ago, passage of the National Environmental Policy Act (NEPA) of 1969 and issuance of the P&S for water resources planning, by the U.S. Water Resources Council (1973), demonstrated the need for evaluation frameworks that consider environmental and economic objectives and tradeoffs with national welfare improvement in mind. Both documents were extraordinarily prescient in anticipating contemporary desires to fulfill the potential of sustainable development.[2] Although the principles and procedures of the P&S are well grounded in decades of water resources planning – representing the most practical elements of decision theory, social choice theory, economic theory and benefit-cost theory—it is only in the past decade that the routine implementation and integration of these principles at the project level was

[2] The preamble to NEPA (Section 101(a), NEPA 1969) lays out the vision of sustainable development as the "… conditions under which man and nature can exist in productive harmony, and fulfill the social, economic, and other requirements of present and future generations…" The P&S, with its four Federal planning objectives, representing National Economic Development (NED), Environmental Quality (EQ), Social Well Being (SWB) and Regional Economic Development (RED) principles anticipated, by 23 years, the principal goals of sustainable development as presented by The President's Council on Sustainable Development (1996).

possible (although not routinely practiced). The P&S framework was modified in 1983, resulting in the P&G that the Corps currently uses in it water resources planning studies.

In addition to needing information to assure the formulation and evaluation of effective projects, the reality of finite investment resources make it necessary to prioritize among projects. Prioritizing numerous similar investments, or the "portfolio problem", requires choosing several worthy projects from among many in which to invest, depending on the goals and objectives for investment. The portfolio dilemma is especially relevant to ecosystem management. Given Federal, state and local budget limitations and considerations of national (Federal) interests and ecosystem management goals and significance of resources, decisions must be made regarding which projects to fund in a given area and in a specified time period. There are numerous "good things to do"; however, we cannot afford to do them all. Choices must be made, and selection should focus on those initiatives that address significant resources and will result in significant outputs in relation to these resources. How to accomplish this evaluation process in a fairly routine and uniformly applied and transparent manner is the focus of this inquiry.

**1.5 Fostering Consistent Theory Across Different Management Decisions**

Improvements in environmental benefits analysis for ecosystem restoration planning should also improve analysis conducted as part of regulatory permits, mitigation planning, and environmental impact analysis. There are numerous perspectives and evaluation philosophies that have been promoted by academicians and that have been adopted by the various agencies to reflect the intent of legislative mandates for their respective programs. For example, within the Corps of Engineers, there are three distinct evaluation philosophies covering virtually the same resource base (e.g., the Nation's waters, including wetlands) within the planning program, the operations programs and the regulatory program. The evaluation principles and procedures are different, as are the decision criteria and rules. Hence, the outcomes of management decisions may vary as each of the evaluation frameworks are applied separately. Fortunately, there are initiatives within the Corps to bring to bear many of the general principles of planning and evaluation; for example, a comparable approach to that advocated in this report has been advocated for use in valuing wetland "services" for the Corps' Regulatory program (King, et al., 2000). There are many resource management agencies throughout the Federal establishment, each with different legislative mandates, creating a vast proliferation of procedures, methods and models all presumably reflecting sound resource management principles and evaluation criteria. The difficulty in harmonizing these different evaluation perspectives has been a substantial obstacle to integrated water resources management, which is perhaps more important than the absence of a truly representative ecological analysis framework.

Progress in environmental benefits analysis will not only improve ecosystem restoration project formulation, evaluation, and justification, it will also improve other environmental analyses and decision making. Ecological theory is the same for all activities that affect ecological resources. However, the extent to which it is applied varies among agency missions, programs, and philosophies both within the Corps and across agencies. Still these various programs (e.g. natural resources management, environmental regulation, and ecosystem restoration) can benefit from, and contribute to improved approaches that infuse contemporary ecosystem science into program objectives and decision-making.

It is essential that efforts to improve environmental benefits analysis proceed via collaborative partnerships with other agencies. While each agency has experience and expertise, none claim the practical, workable construct of ecosystem models and evaluation procedures needed for complex management and investment choices. However, nearly all of the agencies advocate "better science" as important for improved ecosystem management. Some have developed methods or have ongoing work that may be applicable or adaptable to our decision frameworks and analytical needs.

It is prudent and necessary to work with the other agencies: “prudent”, to tap their knowledge and expertise, and to leverage research and development resources; “necessary”, because we want their "buy in" on the methods we propose to use. An underlying goal of this effort is to determine whether it is possible to achieve a common understanding and acceptance of a shared set of methods for ecological analysis among the Federal agencies responsible for natural resources management and/or environmental regulation. Many agencies currently use ecological evaluation methods that are comparable to those used by the Corps, and many have been developed jointly. However, the use of similar ecological analysis tools applied within disparate, non-economic evaluation frameworks would still represent a major obstacle to integrated watershed management or ecosystem management. Coordination of management actions, projects and regulatory decisions would be hampered by the lack of agreement about the relative cost-effectiveness of complementary measures that would be advocated by each of the respective agencies to achieve a set of ecosystem management goals.

The NEPA procedural guidelines encourage the use of bio-economic analysis as part of environmental impact assessments, as does virtually every recent National Research Council report on aquatic restoration actions and watershed planning processes. The Corps, along with several other resource management agencies has developed different approaches to bio-economic analysis. These methods form the basis for additional options for improvements in analytical approaches that might be developed. Box 1.1 summarizes a number of agency and other organizational efforts that may be useful to consider in the development of near-term and longer range approaches for improving environmental benefits analysis. Some of these efforts, along with others we hope to identify as the study proceeds, may be useful in shaping opportunities for collaboration in the development of new methods or refinement of existing methods for improved application in ecosystem restoration planning and other aspects of ecosystem management.

**Box 1.1 Efforts by Others with Potential Applicability to Improving Environmental Benefits Analysis**

**1. USGS - Biological Resources Division** *(FWS (Ft. Collins group))* Adapting HEP to community scale; current target is bird communities and vegetative structure. Some work on oak-based wildlife community.

**2. Forest Service** (Rocky Mt. Experimental Station) Moving away from HEP, greater emphasis on larger scale models that incorporate material cycling and spatially explicit models (e.g. FRAGSTATS). Also, “choosing by advantages” approach, which incorporates: Public Health and Welfare; Environmental Impacts; Project Continuity/Viability ; Legal Issues; Cooperators/Partnerships; Other (e.g. acceptability).

**3. Department of Energy** - Some work emphasizing energy flow and carbon cycling, climate change - models are still in formative stages.

**4. NOAA** - NMFS -Work is underway on a Success Criteria Report which discusses incorporation of structure & function in monitoring criteria; recently published *Habitat Restoration Monitoring Toward Success, A Selective Annotated Bibliography*. *N*OAA- NOS - *Habitat Equivalency Analysis* - developed as part of their Damage Assessment and Restoration Program to assess impacts of spills and other perturbations and to develop compensatory mitigation requirements.

**5. Environmental Protection Agency** - The Office of Research and Development, Corvallis Lab - Synoptic Approach; Wetland Landscape Profiles; Wetland Condition Assessment; Alternative Futures; PATCH, a method which links population models for terrestrial species within a region. Wetland Bioassessment Methodology. Work is underway to develop a bioassessment method for evaluating wetland health for use by wetland a water quality managers. The method builds of the Index of Biotic Integrity (IBI) developed for streams. Some watershed models.

**6. Natural Resources Conservation Service -** information incomplete; method for indicators of wetland functions.

**7. Bureau of Reclamation -** work on “multipurpose” analysis (e.g. recreation and environmental needs); restoration guided by compliance emphasis.

**8. National Science Foundation -** some ongoing work related to biodiversity, food webs, including basin ecosystem productivity algorithms linked with food web processes; most work has a terrestrial focus. Some community model work is underway. May not be emphasizing predictive tools.

**9. The Ecological Society of America -** Report on Ecosystem Services. (Daily, et al); other papers and reports, workshops proposed - academic plus some “communication plans”.

## Section 2. Civil Works Ecosystem Restoration: What, How, and Why?

This section describes the planning framework established for Civil Works ecosystem restoration. Drawing on Corps planning regulations and supporting policy information presented in Box 2.1, it outlines the basic guidance established for Civil Works ecosystem restoration planning, and examines the implications for characterizing and evaluating restoration outputs.

### 2.1 Ecosystem Restoration Defined

Natural ecosystems are self-regulating complexes of habitat and biotic communities, which vary in form and function, while consistently maintaining attributes that differentiate them from other ecosystems. They typically are recognized by the composition of species and population life stages of the communities, and by habitat attributes shaped by the biotic community. For example, a “cypress-dominated forest wetland” is an ecosystem recognized not only by the dominant species, but also by an assemblage of associated species, water-stained brown by dissolved organic matter, sediments rich in particulate organic matter, standing and downed dead woody debris, and other physical-chemical characteristics. The attributes of many ecosystems are disproportionately influenced by one or a few “keystone” species, such as alligators in cypress swamps. Numerous identifiable complexes of community and habitat are associated with the rivers, floodplains, coastal systems, and shore areas influenced by Civil Works activities.

“Ecosystem restoration” is defined by Corps policy documentation as management actions that “attempt to accomplish a return of natural areas or ecosystems to a close approximation of their conditions prior to human disturbance, or to less degraded, more natural conditions.” The first part of this definition suggests that restoration is a concept that relies on some historical record of previous ecosystem condition as a target for management actions. The second part, on the other hand, recognizes that many ecosystems have been altered to such an extent that even partial return to some previous condition may not be possible. Further, there often may not even be any reliable historical record of previous ecosystem conditions that could serve as a target for restoration actions. These factors imply that, whether or not a return to some specific historical ecosystem condition is possible or practical, Corps efforts to restore ecosystems should seek to establish more natural, functioning and self-regulating systems.

### 2.2 Focus of Ecosystem Restoration

Corps environmental management expertise generally centers on the hydrology and geomorphology of aquatic systems. Corps restoration projects typically focus on significant water and related land resources of river and coastal ecosystems, including their associated floodplains, shores, and wetlands. The boundaries of these natural systems typically define the appropriate focus of all Civil Works activities, including traditional flood damage reduction and commercial navigation projects as well as

**Box 2.1. Basics of Civil Works Ecosystem Restoration: Excerpts from Planning Regulations & Supporting Policy Information**

**Ecosystem restoration defined**

"Civil Works ecosystem restoration initiatives attempt to accomplish a return of natural areas or ecosystems to a close approximation of their condition prior to disturbance, or to less degraded, more natural conditions. In some instances a return to pre-disturbance conditions may not be feasible. However, partial restoration may be possible, with significant and valuable improvement made to degraded ecological resources. The needs for improving or re-establishing both the structural components and the functions of the natural area should be examined. The goal is to partially or fully reestablish the attributes of a naturalistic, functioning and self-regulating systems." [EP 1165-2-502, Section 7b]

**Focus of ecosystem restoration**

"Corps activities in ecosystem restoration should concentrate on engineering and other technical solutions to water and related land resource problems, with emphasis on improving degraded ecosystem function and structure. Those restoration opportunities associated with wetlands, riparian and other floodplain and aquatic systems are likely to be most appropriate for Corps involvement. The Corps will focus its restoration efforts on those initiatives most closely tied to Corps missions and areas of expertise. There may be instances where components of ecosystem restoration problems or opportunities are better addressed by other agencies through their missions and programs. Generally, it will not be appropriate for the Corps to implement ecosystem restoration activities on upland, terrestrial sites which are not closely linked to water and related land resources or on Corps project lands." [EP 1165-2-502, Section 7l]

**Objective of ecosystem restoration**

"The Corps objective in ecosystem restoration planning is to contribute to national ecosystem restoration (NER). Contributions to national ecosystem restoration (NER outputs) are increases in the net quantity and/or quality of desired ecosystem resources." [ER 1105-2-100; Section 2.2b] "The purpose of Civil Works ecosystem restoration activities is to restore significant ecosystem function, structure and dynamic processes that have been degraded." [EP 1165-2-502, Section 7l]

**Evaluation of ecosystem restoration**

"Measurement of NER is based on changes in ecological resource quality as a function of improvement in habitat quality and/or quantity and expressed quantitatively in physical units or indexes (but not monetary units)." [ER 1105-2-100; Section 2.2b.] "Ecosystem restoration outputs must be clearly identified and quantified in appropriate units. Although it is possible to evaluate various physical, chemical, and/or biological parameters than can be modified by management measures which would result in an increase in ecosystem quantity and quality in the project area, the use of units that measure an increase in ecosystem value and productivity are preferred. Some examples of possible metrics which may be used include habitat units, acres of increased spawning habitat for anadromous fish, stream miles restored to provide fish habitat, increases in number of breeding birds, increases in target species and diversity indices. Alternative measures of ecosystem value and productivity may be used upon approval by CECW-P. Monetary gains (e.g., incidental recreation or flood damage reduction) and losses (e.g., flood damage reduction or hydropower) associated with the project shall be identified." [ER 1105-2-100, Section 3.5c(1)] ecosystem restoration projects. But whereas traditional Civil Works projects generally rely on management measures to eliminate hydrologic extremes, ecosystem restoration generally requires measures to reintroduce natural hydrologic variability. The key to restoring the attributes of functioning and self-regulating aquatic, wetland and other floodplain ecosystems is the reestablishment of more natural spatial and temporal variability of flow regimes.

In addition, the success of restoration efforts depends largely on how well management decisions incorporate ecological processes outside the immediate scope of projects. For most Corps projects, the physical environmental forces and source materials needed to establish and sustain project success derive from a larger watershed, estuarine, or coastal context. This means that restoration projects should be designed and evaluated within a regional context and with consideration for all factors determining the desired ecosystem form and function.

### 2.3 Objectives of Ecosystem Restoration

Civil Works activities alter the structure and processes of ecosystems. The evaluation of such alterations for decision-making purposes requires a standard of value for indicating whether a change is better or worse. That valuation standard value should follow logically from the stated objective in Civil Works planning.

In traditional Civil Works planning, the Federal objective is defined as utilitarian; that is, to contribute to the satisfaction of human preferences. Economic value provides an empirical "account" of the contribution of Civil Works activities to preference satisfaction, and represents the standard of value specified by Corps guidance for the evaluation of traditional projects. Further, Corps guidance specifies the specific purposes -- or desired economic outputs -- to be served by traditional projects. These economic outputs can be viewed in terms of closely related "ecosystem services".

As depicted in Figure 2, the structural features and ecological processes of an ecosystem--as affected by environmental forces and constraints, management actions, and social and economic activity in the area--yield a mix of functions that in turn provide various "services" valued by society. Ecosystem services have been defined as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Dailey, 1997). As this definition implies, ecosystem services can be viewed as the link between the natural environment and human benefits. That is, the service concept connects an ecological focus on "what ecosystems do" with an economic focus on how ecosystems contribute to the satisfaction of human preferences. As such, the concept embodies both an ecological dimension and a human dimension. Table 2.1 provides a list of example ecosystem services and the various ways in which they can contribute to economic value.

Traditional Civil Works purposes include many of the production and consumption activities listed in the right hand side of Table 2.1, and these in turn are closely linked to

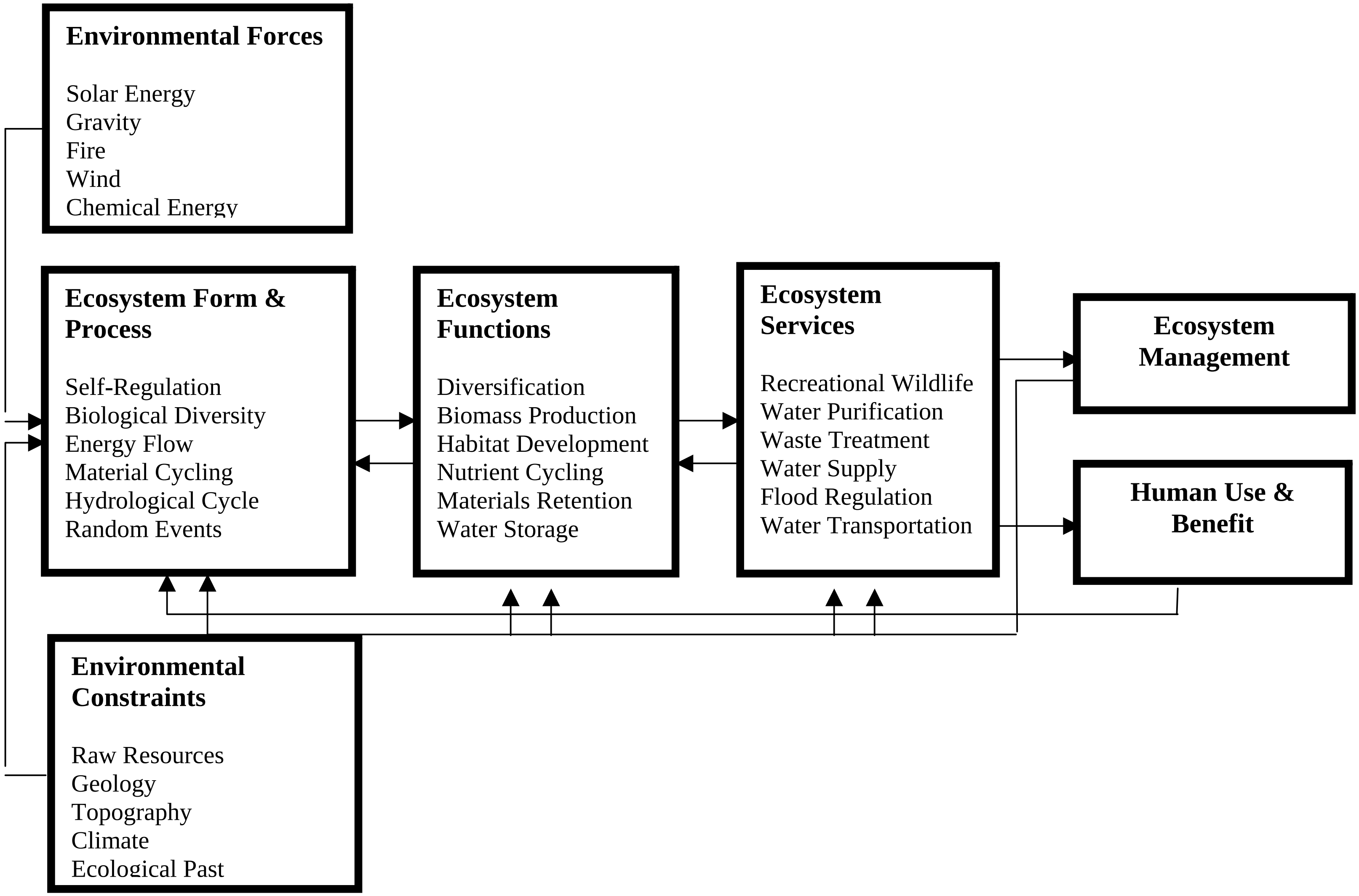


**Figure 2.1 General relationships among ecosystem form and process, functions, services, and human uses and benefits in river and floodplain ecosystems.**

**Table 2.1 Examples of Ecosystem Services and Associated Human Uses & Benefits**

<table>
<tr><th>Ecosystem Services</th><th colspan="2">Channels Through Which Ecosystem Services Contribute to Economic Value</th></tr>
<tr><td rowspan="3">• Disturbance Regulation (flood, wind & wave)<br>• Waterway Transportation Links<br>• Water Storage<br>• Water Purification<br>• Sediment Trapping<br>• Waste Treatment<br>• Biological Pest Control<br>• Climate Regulation<br>• Rare and Unique Species/Genetic Store<br>• Wildlife Support (e.g., food chain, nursery)</td><td>Direct Passive Use</td><td>• Personal satisfaction derived from the knowledge that rare ecosystems & associated functions & services are intact, independent of any actual or anticipated active use</td></tr>
<tr><td>Direct Consumption</td><td>• Community Flood & Storm Protection<br>• Municipal & Residential Water Supply<br>• Consumptive & Non-consumptive Recreation<br>• Aesthetics, Observation & Study</td></tr>
<tr><td>Production Inputs</td><td>• Land Productivity for Agriculture<br>• Commercial Navigation<br>• Hydroelectric Power Generation<br>• Water Input for Agriculture & Industry<br>• Commercial Fishing, Hunting/Trapping, etc.</td></tr>
</table>

one or more underlying ecosystem services listed on the left side of the table. Indeed, it is these associated ecosystem services that are the focus of plan formulation for traditional Civil Works projects. So, for example, commercial navigation projects focus on intensive enhancement of natural waterway transportation links, and flood damage reduction projects focus on enhancement of the natural flood regulation service

While the relationship between ecosystem services and the traditional Civil Works objective and specific purposes is straightforward, the relevance of services to the ecosystem restoration mission is not as apparent. Corps planning regulations and supporting policy documentation specify that the Federal objective in ecosystem restoration is to increase the net quantity and/or quality of desired resources through the restoration of significant ecosystem function, structure and dynamic processes that have been degraded. The relevance of ecosystem services for the restoration mission depends on how this objective statement is interpreted in terms of desired ends. At least two possible motives for movement along a restoration gradient can be identified.

First, restoration might be sought purely for utilitarian reasons, implying a concern for services that people value. That is, management actions might seek to restore the hydrologic conditions thought necessary to secure a mix of ecosystem services and associated human benefits only because that is the best plan for reestablishing deficient services. But, when services ordinarily associated with a more natural condition are better gained by artificial means, a simulation of natural measures—a naturalistic approach—or even a highly artificial plan, might be chosen. The approach to restoration might be called “the manager knows best” approach and is based on careful analysis of resource and service flow from a variety of natural and artificial management measures. As indicated in Table 2.1, natural ecosystem services can produce economic value in a variety of ways. In the extreme, people may derive satisfaction from the mere knowledge that rare ecosystems and associated services are maintained in good condition. Such assurance is said to produce “passive use value” that is independent of actual or planned visitation or active use. By contrast, “use value” is generated when people actively use ecosystems services by consuming them directly or indirectly as inputs into commercial production. For example, restoration can augment water purification and wildlife support services that generate use value directly by improving recreation opportunities, and indirectly by supporting commercial fisheries. Restoration of nature’s services can also generate use value in more subtle and indirect ways by supporting general economic and social activity—for example, services such as climate regulation, sediment trapping, and waste treatment support and prevent damage to a wide range of consumption and production activity. Of course, restoration in any context would not be expected to augment all potentially affected services—the flows of some natural services would likely decrease as others increase. And inasmuch as restoration involves movement towards greater hydrologic variability, certain natural services might be served at the expense of other services that previously had been enhanced to serve to traditional Civil Works purposes.

A second possible motive for pursuing restoration is the “naturalness” of ecosystem hydrology and geomorphology, as an end in itself (Shabman, 2002). This approach is not independent of the resulting mix of service flows, but assumes that whatever results ecologically is more acceptable than the results from any other alternative. This might be called a “nature knows best” approach. It ignores the service flows from proposed management measures based on the *a priori* judgment that no better plan alternative exists. On the surface at least, this seems to be the position of some environmental interest groups that advocate a return to free-flowing rivers in certain contexts. The notion that naturalness is an independent value to be advanced by civil works planning is at odds with the traditional civil works objective rooted in utilitarianism, but represents one plausible motivation for Corps restoration activities.

Acceptance of that interpretation does not imply that Civil Works restoration is unconcerned with the interests of people, however. There exist theories of value that recognize human-based values as distinct from utilitarian value. Perhaps most notably, “Kantian Ethics” asserts that human society can establish moral rights and obligations that recognize the value of certain things and outcomes as ends in themselves (National Research Council, 1999b). In the restoration context, a Kantian perspective might assert that in some cases ecosystem naturalization is the “right thing to do” for humanity (not for plants and animals, which are given no moral standing), and should be promoted as a matter of collective responsibility rather than individual preference satisfaction.

Corps planning guidance does not specifically establish the desired ends of restoration as naturalness for its own sake, or for supporting natural ecosystem service outcomes. Instead, guidance emphasizes the “significance” of resources and effects for judging the desirability of restoration. The significance concept is defined in terms of institutional, public or technical recognition of importance, and as such seems broad enough to include both naturalness and associated services as desired restoration ends. As one example where both types of value may be relevant, consider the plan now being pursued to restore a portion of the historic Florida Everglades system. In this case, restoration of a more natural pattern and timing of flows (and whatever ecologic response that results) might be viewed as a valued end in itself, and also as a necessary condition for improving ecosystem services that give rise to utilitarian value.

### 2.4 Evaluation of Ecosystem Restoration

Corps regulations specify that restoration outputs must be evaluated in non-monetary metrics, with preference given to “units that measure an increase in ecosystem value or productivity” (see Box 2.1). Ideally, these value metrics should follow from the desired ends of restoration in any particular context. For example, if restoration of naturalness represents one valued end to project stakeholders, then the non-monetary metrics chosen for evaluation might be derived from the pre-disturbance ecosystem condition or some related reference condition. On the other hand, if the restoration of ecosystem services that give rise to utilitarian value is a prime concern, then stakeholder may demand project evaluation metrics that indicate the desired direction of change in one or more services. When services and associated utilitarian values are one project goal, the best indicator of the output significance is monetary benefits. However, natural ecosystem services largely represent “public goods” that provide benefits that are collectively supplied to all potential users, and thus are not traded and priced in the marketplace like private goods. As discussed in more detail in Section 6, the lack of market prices for natural ecosystem services is a significant barrier to economic valuation of changes in service outcomes resulting from restoration. This may at least partly explain why Corps regulations require restoration outcomes to be evaluated in non-monetary terms. At any rate, to the extent that one goal of restoration is to augment valued services, then project evaluation requires planners to move beyond metrics indicating a more natural state to non-monetary metrics that indicate the desired direction of change in desired service outcomes.

# Section 3.  Ecological Concepts Underlying Environmental Benefits Analysis

## 3.1 Section Objectives and Background

### 3.1.1 Objectives

The objective of this section is to summarize ecological concepts of potential relevance for characterizing and evaluating ecosystem restoration outputs.  It considers the types of ecosystem outputs and indicators of environmental benefit that might be useful for Corps plan formulation and evaluation, including the possibility that there may be some inclusive non-monetary measure of environmental benefit that may have wide applicability for ecosystem restoration planning. In addition, it addresses the scale and character of natural ecological inputs from the influential ecosystem environment (the system context) that generally determine the ecosystem structure and functions that need to be considered for predicting ecosystem outputs. A secondary intent is to provide a *conceptual basis* to aid in the selection and development of physical and mathematical models useful for plan formulation and evaluation of ecosystem outputs indicative of environmental benefit.  Relevant model types are discussed in Section 4.

The discussions within this section include:

- Corps policy that contributed to determination of ecological concept relevancy.
- Ecological concepts, beginning with *ecosystem structure and function*, which is the focus of Corps restoration purpose and definition of *ecological resources*.
- The concept of *natural ecosystem service*, as conceived primarily by ecologists and ecological economists, and attempts to describe "*naturalness*" in the concepts of *natural and cultural integrity*, and related concepts of *ecosystem health* and *sustainability*.
- The broadly stated concepts relevant to output characterization that are pertinent for restoration formulation and evaluation, including *biodiversity; ecosystem self-regulation, resilience* and *sustainability; ecosystem production* and *biomass*; and *ecosystem materials flow and cycling* (with hydrologic cycling as a special case).
- An important culminating discussion of the often different responses of ecosystem function and structure to natural and managed restoration process leads into a summary discussion of the *roles of local and global biodiversity* as benefit indicators for plan formulation and evaluation.
- A brief description of the character and scale of ecosystem inputs necessary for consideration in comprehensive formulation and evaluation methods and models.

### 3.1.2 Policy Indicators of Ecological Concept Relevance

Many ecological concepts most relevant to environmental benefits analysis in Corps ecosystem restoration planning are indicated by Corps policy.  Much of the relevant policy has been summarized in Chapter 1 and 2. A few additional points are summarized here.

The concept of environment Corps policy limits *evaluation* of environmental improvement from ecosystem restoration to *ecological* resource quality. It clearly excludes cultural and aesthetic attributes of the environment as it is more inclusively defined in the P&G (WRC 1983). Moreover, the ecological resource quality to be considered "is a function of improvement in habitat quality and/or quantity". The concept of habitat in Corps policy is defined by the needs of living inhabitants—that is, the inhabitants comprise the resource quality generated by habitat improvement. Thus the ecological indication of resource quality is found in the inhabitants—the species and communities—not the habitat itself. Thus the Corps formulates for habitat as defined by the needs of the inhabitants and evaluates plans based on the confidence that the habitat will become inhabited once it is provided. This is the Achilles heel of many existing planning methods and models.

While habitat improvements may affect non-living outputs from the ecosystem (e.g., water supply, water quality, carbon dioxide emissions, sediment export), they are not among the significant resources that justify a restoration investment. However, the responses of nonliving outputs to restoration also need to be considered for their effect on the total benefit realized by restoration plans. The completeness with which this is done may determine the degree of concern associated with the NRC (1999a) fear that habitat-based methods, when used alone, may fail to consider all of the national interests.

The living resources targeted for ecosystem restoration should contribute to the "net quantity and/or quality of desired ecosystem resources" both "in the planning area and in the rest of the Nation". Thus the scope of planning method consideration extends to the entire ecosystem condition in the U. S., not just the local fraction of the ecosystem existing in the project area or environs. Because local sites in ecosystems often express widely different attributes from much of the ecosystem, the larger perspective is important for determining the degree of human effect and resource degradation that has occurred in the ecosystem. Ecological resources may be locally scarce, but nationally abundant. The national perspective sets a standard for judging the scarcity of ecosystem resources, which is an important consideration for determining its social significance.

The outputs from ecosystem restoration plans are to indicate a *significant* change in *significant* resource condition to a *less degraded* and *more natural condition.* "Restored ecosystems should mimic, as closely as possible, conditions which would occur in the area in the absence of human changes to the landscape and hydrology". The term landscape refers to the full set of surrounding ecosystem conditions that influences the project ecosystem condition. Another intent is "to partially or fully reestablish the attributes of a naturalistic, functioning, and self-regulating system" to assure as long as possible the long-term continuity of improved resource condition. Thus, whatever more natural (or naturalistic) condition is established in support of significant living resources, the ideal condition is functionally self-regulating.

To help planners focus on the remote as well as proximal influences determining self regulation and long-term persistence in the project area, including the entire community-habitat complex, policy emphasizes the importance of viewing the project area as a dependent subsystem in a larger systems context. “Ecosystem restoration projects should be formulated in a systems context to improve the potential for long-term survival of aquatic, wetland, and terrestrial complexes as self-regulating, functioning systems.” The Corps Environmental Operational Principles reinforce this notion of long term continuity or beneficial results, and introduces related concepts: “Strive to achieve environmental sustainability: An environment maintained in a healthy, diverse and sustainable condition necessary to support life”. The closely related ecological concepts of *ecosystem integrity* (including both natural and cultural integrity) and *biodiversity* pertain especially to policy concepts of naturalness and *sustainability*, and the concept of ecosystem *self-regulation* is an especially critical master-function.

It is possible, if not likely, that in some cases, a more natural condition (whatever results from removing human effect) is in itself the ecological output of significance. According to Corps policy, that increased naturalness needs to be reflected in the living organisms comprising the significant resources and the habitat through which those resources are to be restored. Based on past restoration motivations, however, increasing the naturalness of the habitat-community complex may not regain specific resources of significance, especially when the restoration is only partial and the targeted resources are among the rare species in the ecosystem.

The existing understanding of ecosystems described in this section suggests that common conceptual and mathematical models of ecosystem naturalness will most confidently predict reestablishment of all ecosystem resources of significance only when full restoration of a natural state is achieved throughout the ecosystem. The concept of ecological *resilience* is especially relevant to this judgment because of what it has to say about differential responses of function and structure to natural or engineered restoration, depending on system context, degree of alteration, and intensity of stress. This issue is critical because human effects are so pervasive and persistent, in large part because they are desirable effects, that restoration to a fully natural ecosystem state is improbable at best in most ecosystems.

Policy identifies a number of ecological concepts of high relevancy to environmental benefits analysis for ecosystem restoration projects. These include the interrelated concepts of natural ecosystem *structure, function, dynamic process, ecosystem integrity* (both natural and cultural*), biodiversity, self-regulation, resilience, functional stability, functional redundancy, sustainability, ecosystem health, production, materials cycling* (including the *hydrologic cycle), landscape* and related ideas. To the extent that these interrelated concepts can reflect the effects of human impacts on ecosystems and the effects of restoration on human perceptions of significant change, they may be considered as important attributes of *environmental quality* associated with ecosystem naturalness and resources of significance. Some of these ecological concepts are more thoroughly developed than others. While many questions remain about concept validity and practical applications, the sum forms a theoretical basis for the formulation for and the evaluation of ecosystem restoration benefits.

### 3.2 The Ecosystem Source of Human Service

The concept of ecosystem service is a useful entry way to defining the relevancy of ecological concepts to environmental benefits analysis. Ecological function and structure are the traditional subjects of ecological investigation, but their relevancy to society and public policy often gets lost in the science. The combined growth in human population and per capita human effect is rapidly changing natural ecosystem function and structure, with potentially threatening consequences that continues to concern many ecologists and some social scientists. To bridge the gap between ecological science and policy applications, a growing group of ecologists and social scientists (Daily 1997, Daily et al. 1997) have developed a concept of natural ecosystem service to humanity. This recent development builds on a long history of renewable natural resources management based in ecological science and resource utility. With respect to the connection between ecosystem function and service, Daily (1997) states:

> "In addition to the production of goods, ecosystem services are the actual life-support functions, such as cleansing, recycling, and renewal, and they confer many intangible aesthetic and cultural benefits as well".

In this view natural ecosystem services are those ecosystem *functions* that confer both tangible and intangible benefits to humans. While this definition seems to equate natural function with natural service, service is a *social concept*, based on the wants (usually recognized by society at large) and needs (often recognized only by a subset of specialists) of society. Ecosystem function, in contrast, is a service-neutral ecological concept. Based on the general acceptance of much human modification of ecosystems to serve humanity by totally replacing or enhancing preexisting natural functions, only a subset of natural functions *significantly* contributes to human service. By implication, a substantial fraction of the earth's natural ecosystem function was redundant and its service to humanity could be and was improved as revealed by a net gain in human welfare. Daily (1997) and Daily et al. (1997) emphasize, however, that as the impact of human kind on its environment has escalated, much more of the remaining natural function of ecosystems significantly contributes the remaining natural resources and services of substantial significance, and some significant contribution has already been lost.

Greater naturalness may, in itself, be the service of significance recognized not so much by the removal of human effects causing a deficiency of specific services, but by the removal of the effector (a dam, levee, channel, sea wall etc). This perception of naturalness does not require any past or present reference conditions to model in restoration process, it simply requires removing the effector. It is not an ecological concept (no vision of ecological change is involved), but rather a social concept of naturalness independent of response in the material world. The service value derives from the degree of dissatisfaction perceived in the edifice to be removed. The significant service is realized immediately upon removal of the effector (including any human evidence of the removal process itself). What comes of it in ecosystem function and structure is irrelevant. There are no material natural resources of significance from which the service originates and no service flow to resource utility. Because the service is not provided by the resulting material world there is little need for ecological/environmental methods and models to formulate for outputs and evaluate effects.

While this service may be socially significant, Corps policy seems to preclude its consideration. Corps policy is based in ecological concepts of naturalness, concepts based in the material world. Corps policy clearly indicates that the resource quality contributing to NER is to be determined through degraded ecological resources of social significance that respond positively to habitat restoration. Removal of perceived effectors is a restorative action, but not a vision of a restored condition. Although removal of human edifices, regardless of material outcome, may provide a valued service and a motivation for seeking Corps actions, the Corps determination of investment worthiness stems from the services conferred by specific manifestations of material resources. Whether and how the value of this non-utilitarian service can be judged, it would be judged incidental to the restoration of socially significant living resources. Any other non-utilitarian service not grounded in material ecosystem change would require similar consideration.

Conceivably, naturalness may be viewed as a collective material resource providing significant, but non-utilitarian service that results in intangible benefits. Different from value held in the removal of a human edifice, this recognition of significance in naturalness is held in specific manifestations of ecosystem structure and function. What comes of the edifice removal (or other alternative restorative action) is important result, not the edifice removal per se. This concept of naturalness is more likely to be based in indicators of reference-ecosystem resource condition. Any utilitarian concept of collective naturalness, such as for nature observation, is also based in specific manifestation of ecosystem function and structure. In other cases only a subset of significant services identifies the natural function and structure that comprise the underlying significant resources and naturalness is more of a means to an end than an end in itself. These services are most likely to be utilitarian, but non-utilitarian services are also conceivable.

Whether or not the social and ecological concepts of natural function are always or ever identical is uncertain and may be critical to realizing restoration objectives based on the material outputs amenable to scientific measure. The ecological concept of naturalness is based in scientific measure of human effect on the material condition of ecosystems, the same material conditions that comprise the resource structure and the functions underlying services. While the benefits to humans do not have to be based in material utility, either passive or active, ecological science is limited to the tangible world having physical existence, and how it responds to management. That is, ecological resources may provide services with intangible benefit, but they must somehow link back to tangible properties in the ecosystem if management for those properties, including restoration, is to *predictably result* in desired outputs. This section primarily addresses the ecological-evolutionary concepts of naturalness and resources.

The concept of natural integrity has emerged over the past two decade to ecologically characterize the naturalness of ecosystems.   Less known, but more relevant to the idea of humanity living harmoniously with nature are the concepts of *cultural integrity* and *ecosystem health*, which attempt to provide a theoretical basis for judging the appropriate mix of enhanced ecosystem service and natural ecosystem service. These concepts of natural and cultural integrity, and ecosystem health are closely related by  concepts of resource and service *sustainability*.  All of these concepts are addressed first in this section to establish a foundation for the following discussion of *ecological outputs* most appropriate for indicating *environmental benefits*.

### 3.2.1 Natural Ecosystem Resources and Services

#### 3.2.1.1 Structure, Function and Dynamic Processes

Odum (1962) was the first to clearly describe the interdependent concepts of ecosystem structure and function, and later (Odum 1993) indicated that they formed the foundational resource for sustaining renewable natural resources. The general relationship among ecosystem structural and functional resources and natural services is represented in Figure 2.1 in Section 2. Parallel general examples of these relationships that might be associated with Corps restoration projects are provided in Table 3.1. Although they may appear straightforward, there is actually tremendous complexity in the linkages among ecosystem structure and functions that underlie services (Jorgensen and Muller, 2000). Ecologists have devoted substantial efforts to organize this complexity into manageable and holistic concepts of ecosystem structure (form) and function.

Definitions vary, but most agree that ecosystem function is what the community-habitat complex “does” when it is energized and structure is its material form. Function is process that predictably organizes materials into ecosystem structure, including physical features and species composition, relative abundance, and demographic attributes. Ecosystem function may be primarily physical, as it is in the hydrologic cycle, or primarily biological, as in the processes of population dispersal and ecosystem colonization.  But in all cases both physical and biological form and process interact through the numerous links between habitat and inhabitants.

Ecosystem process is sometimes equated with ecosystem function.  While all function is process, we separate function from other dynamic process.  Ecosystem functions require driving forces that originate from the ecosystem environment, such as the energy in solar radiation, chemical reactions, gravity, and tidal effects.  Most of the driving processes are dynamic (gravity being the major exception) and quite predictable at the source (the sun and moon, and the earth’s chemical composition and mass).  However, random events also are dynamic processes that are not predictable for specific times and places and often influence driving forces through climatic and geological variation. Random events are far from irrelevant to ecosystem function, however, because they are common in ecosystems and interfere with the predictable organization of materials into ecosystem structure. Random events cause residual uncertainty in ecosystem output response to management or to natural events once the predictable relationships among ecosystem properties are understood. Many natural resource management actions have been put in place to ameliorate the effects of naturally random events such as flood, storm, and fire. But many other more subtle random processes influence the biological process of restorations as well, especially with respect to species recolonization of disturbed ecosystem sites.

Structure is the spatial arrangement of living and nonliving materials in an ecosystem at any one time and sequentially through time. The bio-structural components of ecosystems are created, maintained, linked, and destroyed through genetically coordinated function and function is maintained through structural dynamics. To refer to one implies the other. Physical mass and its instantaneous distribution in its various forms are measures of structure. For example, standing-crop biomass is one measure of ecosystem material form and biomass production is function. Similarly, stream discharge can be a function of ecosystems while water mass is its material form. Structure is sometimes referred to as the elements of ecosystems, especially in landscape ecology. As the term is used here, structure includes the arrangements of elements (individual organisms and other physical objects) in space.

Ecosystem structure, function, and dynamic process occur in all ecosystems regardless of how modified they may be by human actions. Whether or not structured or otherwise modified by humans, all ecosystems conform to fundamental laws of physics. A humanly engineered form of ecosystem structure functions to deliver the energy and materials needed by society according to the same natural laws that the sun delivers solar radiation to plant photosynthesis. All things human and nonhuman are natural in the context of natural "law" and neither good nor bad. Maximum human welfare, which is defined to be "good" or desirable, lies at that optimum condition somewhere along a gradient of human effect in a fully natural world. Thus the concept of natural function and structure as it has been defined in Corps policy—as occurring only in the absence of human effect—fits more comfortably into philosophical knowledge than scientific knowledge. What is most meaningful here for resource management is not whether human effects are natural, but what in ecological and evolutionary science is most meaningful for assessing human effects on ecological function and structure that somehow relate to the satisfaction of human wants.

Expression of ecosystem structure and function is often characterized by *diversity*, which is the variation in form and function that occurs in the genetic makeup of individuals and populations comprising a species, among aggregates of species within ecosystems, and in landscapes including numerous ecosystems. All diversity that is influenced by biological process in ecosystems has become known as *biological diversity*, or, more commonly, *biodiversity*. Landscape-level diversity is determined by the arrangements of different habitats and communities, including size, edge to area ratios, connectivity and patterns of habitat and community distributions with respect to other ecosystems in the landscape. Preserving genes, species, and even entire communities may be insufficient if *the landscape context* of community and habitat does not also provide the proper environment and supplies of energy and material for organizing life function and structure.

### 3.2.1.2 Roles of Genetic Information, Biodiversity and Species Composition

The expression of *genetic information* held in ecosystems is often identified as the most basic manifestation of ecosystem structure and function because it is the architect for other ecosystem structure and function at all hierarchical levels (Haywood 1995). The genetic information in ecosystems is most typically indicated in the biodiversity expressed in species and communities. Thus biodiversity indicators of genetic diversity and relative scarcity show potential as an indicator of environmental resource value once their expression is matched with indicators of human service.

Given the general ecosystem setting in which natural communities have evolved, genetic information generally determines the biomass and production of whole biotic communities through the collective function and structure of all species adapted to the ecosystems. Genetic information is transferred forward to successive generations of species populations making up the biotic communities of ecosystems. The functional and structural interactions within and among ecosystems start at the level of molecular events and work up through tissues, organisms, populations and communities interacting with their physical habitats. At the ecosystem level, the myriad miniscule structures and functions are "bundled" into conceptually more manageable aggregates such as emergent herbaceous vegetation, planktonic herbivores, primary production and carbon cycling.

Despite the complexity of ecosystems, the most commonly encountered measure of diversity is *species richness*—the number of species in a defined area—because it is relatively easy to measure. Also, species richness often correlates with more complex multi-criterion measures of diversity. The relationship between species richness and ecosystem functions, such as *primary production* (the first-level production most often associated with photosynthesis), has been a topic of active research in recent years. The results of this type of research are of exceptional interest to restoration practitioners because of the potential for species richness to serve as an indicator of functional status of ecosystems and, indirectly, as an indicator of ecosystem service.

The relative contribution of each species to structure and function is far from uniform, however, and simple biodiversity indexes can misrepresent exceptional contributions. Just as the different indicators of human physical condition are weighted according to their health implications (e.g., cancer verses acne), each species in a diversity index varies in its importance as an indicator of ecosystem condition. One shortcoming of a species richness index is its inability to discriminate the differences in dependency of ecosystem functions on single species and groups of species. Consistently rare species may invade and exit communities without much noticeable change in ecosystem function and structure. The comings and goings of rare species and their influences in the ecosystem may be below the limits of our ability to detect them in a sampling scheme, given technical and economic limits.

On the other hand, ecosystems can change dramatically when exceptionally influential dominant or keystone species come and go (Paine 1966, Power and Mills 1995).

Keystone species contribute disproportionately (with respect to their abundance) to both the functional and structural integrity of ecosystems, as do species that dominate because they are abundant. Except for keystone species, rare species contribute little to function but equally to simple measures of structural diversity. The potential roles of species richness, dominant species, and keystone species in development of decision-support tools are discussed later in Section 4.

The development of the ecosystem concept has emphasized structure and function, and their relationships. The structure theme typically has highlighted community diversity and composition (e.g., Pimm 1991). The function theme typically has emphasized energy flow through food webs, biomass production, and material flows and cycles (e.g. Odum 1984 , Ollinger et al. 1998, Bartell et al. 1999). Other theory has attempted to integrate the two themes through links between structural diversity and the stability of production and other functions (Pimm 1991, Holling 1996). Hannon (1973) developed the concept of structure through the food-web interdependence of species. He characterized community structure as a changing cross-section of community energy flow through food webs. Golly (2000) concluded that "ecosystem structure is the network of interactions between components of the system". Both structure and function contribute to the natural resources in ecosystems. Structure is the store of resource at any one time and function includes the production and decomposition resulting in the net store of natural resources.

#### 3.2.1.3 Natural Service

Ecologists and economists have identified numerous examples of natural ecosystem service (e.g., Barbier et al, 1995, Daily 1997, Daily et al. 1997, Costanza et al. 1997, Table 3.1). The ecological view is consistent with the discussion of services in Section 2, but emphasizes more the connections between natural ecosystem service and ecosystem function and structure. In this regard, a helpful concept addresses the distinctions between the service recognized directly by the public at large and the service recognized indirectly through the specialized knowledge of ecosystem structure and function. When service is easily recognized, such as provision of watchable wildlife, ecologists are not needed to determine that a service indeed exists. On the other hand, when services are recognized only indirectly by ecologists working their way back from evident impacts on

**Table 3.1. Generic examples of natural ecosystem structure, function, and service. They are associated to various degrees and form with river, coastal, floodplain and other ecosystems managed by the Corps.**

| ECOSYSTEM RESOURCES | | ECOSYSTEM SERVICES |
|---|---|---|
| ECOSYSTEM STRUCTURE | ECOSYSTEM FUNCTIONS | |
| Carbon dioxide; biomass, water area | Thermodynamics; carbon cycle | Climate Regulation |
| Vegetation, floodplain & barrier islands | Wind, wave & flow alteration | Storm and flood Moderation |
| Lakes, ponds, aquifers, ice, biomass | Water retention and delivery | Water Supply |
| Particle size, root mass, debris dams | Soil and sediment movement | Control sedimentation |
| Biomass, sediment, humus, | Material trapping; decomposition | Contaminant removal |
| Species composition and diversity | Predation, disease, competition | Biological pest control |
| Biomass, air, water, species diversity | Plant and animal production | Food production |
| Wood, humus, clay, shell | Production of raw materials | Materials supply for commodities |
| Global species richness | Diversification and life support | Sustained genetic information |
| Water, wildlife composition, topography | Water flow; life process | Recreation/aesthetic |
| Landscape patterns of ecosystem form | Recovery after disturbance | More sustainable service |

human welfare, then public policy needs to be informed about the connection. It is this scientific recognition of service that is most important to Daily (1997) and Daily et al (1997) to demonstrate—because it is not obvious.

Because ecologists deal with the tangible, material world, it is much less likely that ecologists will recognize an intangible human need. It is much more likely that they will reveal utilitarian services than non-utilitarian services. A good summary of a closely related concept can be found in Goulder and Kennedy (1997) in their utilitarian discussion of direct and indirect use of resources. While resources may not have to be used, either directly or indirectly, for services to be recognized by the public, ecologists are not likely to be able to help them out tracking back to ecosystem functions and structure if their is no connection through physical use, including any passive but satisfying sensual perception of the material world. Thus the environmental benefits that are addressed by ecologists are limited to benefits from utilitarian service.

Many uses of natural resources are direct and marketable; associated commercial and recreational services are priced, such as for the prices paid to gain access to and harvest timber and waterfowl from private wetlands. The price is paid because the service and its quality are readily perceived by the users. When natural resources are closely linked to a specific geographical area, that space can be valued indirectly through valuation of the functions (services) associated with direct use. For example, the indirect value of a forested wetland functions that generate timber supply (a recognized service) is relatively easily determined through lumber prices and harvest costs, which indirectly determine timber value, which, in turn, determines the wetland value for timber production (with additional knowledge about production rate and quality, and future demand for lumber). The property values for that specific service are readily determined based on projected logging income because the timber is literally rooted in the wetland and its production rate can be reliably calculated.

The value of other ecosystems contributing to resource harvest is obscured by incompletely defined ecosystem process and boundaries. Natural wetland support services for offshore commercial and sport fisheries production and harvest are much harder to value because services are dispersed and are difficult to tie to a specific area. Private property value associated with commercial fish production hardly exists outside fish-farm pens because most of the resources of value, the harvestable fish, disperse to public waters beyond the control of the property owner. While it is typically not feasible to trace the fish sold in individual economic transactions back to specific wetlands, it is feasible to estimate an average or aggregate service value for sustaining fisheries via backtracking through ecological food webs, fish migration pathways, and various material transport pathways to a general type of wetland condition.

As crude as this approach has been, this type of indirect valuation exercise, working back from direct service value to indirect service value through ecological pathways, has been used in part to justify public protection of coastal wetlands through state and federal permitting procedures. However, understanding of the natural ecosystem structure, function and other process linking to the priced resource is necessary before any estimate can be made of supporting ecosystem service value. There has been a long history of such analysis and decision in Federal and State waterfowl management, and to a lesser extent, other wildlife management. Starting in the 1930s, government wildlife agencies began to buy up lands to restore or create habitat for waterfowl using revenues from duck stamps bought by hunters. The buyers had to sort through land prices to determine the best buys based on the anticipated return in waterfowl-based benefits. Ecological methods were crude, but generally effective, long before models and computers allowed more sophisticated evaluation.

Restoring or setting aside existing habitat for an endangered species is also based on a scientific assessment of the ecosystem structure and functions required to sustain an endangered species. The habitat has no value, however, without the inhabitants. Thus habitat protection and or restoration have to be completely assessed ecologically, including all recovery pathways necessary, or the restoration could prove valueless for the intended purpose.

#### 3.2.1.4 Relating Social Significance to Ecological Concepts

As outlined in Section 2, Corps regulations specify that restoration outputs should be characterized and evaluated in non-monetary metrics that are indicative of institutional, public or technical recognition of resource significance. Institutional indicators most obviously take form as environmental laws such as the Clean Water Act and Endangered Species Act, which emphasis recovery of and sustained maintenance of clean water and rare species. Both attributes of aquatic ecosystems are closely associated with the integrity of naturally functioning ecosystems. Public indicators of resource significance, led by the environmental NGOs, usually emphasize a sustainable ecosystem condition (increasingly referred to as ecosystem or environmental health) in support of human health, rare-species, recreational use, and other sustainable uses with mixed enhanced and natural services.

Technical assessments of significance have been captured comprehensively in statements such as the committee report of the Ecological Society of America about the scientific basis for management of the Earth’s resources and maintenance of life-support systems

(Lubchenco et al. 1991). This professional society identified three particularly critical "problems facing humanity", including: "global change, maintenance of biological diversity, and the sustainability of natural and managed systems." These problems are linked to concerns of global proportion that may lead to resources of global significance.

In response to these problems, the Ecological Society of America has recommended major research initiatives to determine how *ecological complexity* controls global process change (including climate, patterns of land and water use, and environmental chemistry), how *biological diversity* (at genetic, species, and ecosystem levels) controls and responds to ecological process (such as energy and material flows through and between ecosystems), and how to restore and manage ecosystems to enhance *ecosystem sustainability*. The Ecological Society of America also has initiated integration of economic and ecological principles into a concept of natural ecosystem services, and extended the result to decision makers as an issues statement (Daily et al. 1997). The heavy emphasis of technical input on research needs reveals the uncertainty that exists with respect to how consistently the evolving principles and prevalent concepts about ecosystems apply to specific conditions.

Weaving throughout these institutional, public, and technical indicators of ecosystem resource significance is concern over how much alteration the *natural integrity* of world ecosystems can absorb before costly unsustainable states of desirable natural resource condition result. However, the concepts of natural integrity, ecosystem complexity, biodiversity, and sustainability have proven easier to address in the abstract than in practical application. The next subsection summarizes prevalent concepts pertaining to the natural integrity of ecosystems and how it relates to biodiversity and ecosystem sustainability.

### 3.2.2 Ecosystem Integrity

#### 3.2.2.1 The Concept

Standing out categorically among ecosystem concepts of potential output importance is the *natural integrity* of functions and structures with respect to biodiversity maintenance, energy-flow, material-flow, and self-regulating sustainability. The concept of natural ecosystem integrity provides a theoretical basis for *measuring the naturalness* of ecosystems. The concept of natural ecosystem integrity has emerged most fully over the last two decades in response to management mandates, such as those included in the Clean Water Act, which seeks the restoration and maintenance of the physical, chemical, and biological integrity of the Nation's waters. In the narrow sense defined by Angermeir and Karr (1994), ecosystem integrity is the relative completeness of natural ecosystem function, structure, and associated complexity determined by ecosystem evolutionary history, which is reflected in the system's "ability to generate and maintain adaptive biotic elements through natural evolutionary process".

In the sense of the commonly accepted definition of Karr (1981,1991) and Angermeir and Karr (1994), natural integrity pertains only to the completeness of ecosystem structure and function within a specific ecosystem. In practice, natural ecosystem integrity is defined by reference to the state of existing unimpaired parts of ecosystems, and, much less commonly, by reference to a record of some previous more-natural state at the restoration site.

This concept of natural integrity is not universally accepted, however. Ecological progress in finding the "right" definition of ecological integrity has been slow according to others (Barkmann and Windhorst 2000). An important issue is the measurement of ecosystem integrity Ecosystem integrity has been measured using the component parts making up ecosystem structure (Karr 1993) and, less commonly, by using ecosystem functions (Schneider and Kay 1994). Whichever model/method is used, all measurement is based on sampling ecosystem attributes along a gradient of naturalness from most natural to most humanly modified.

A number of models of relative naturalness have been developed based on structural attributes including the Index of Biotic Integrity (IBI; Karr 1981, Karr et al. 1986), the Wildlife Community Habitat Evaluation (WCHE; Schroeder 1996a and b), and the Riverine Community Habitat Assessment and Restoration Concept (RCHARC; Nesler et al. 1995). The most widely known model addressing the naturalness of ecosystem function is the Hydrogeomorphic Approach (HGM; Smith et al. 1995). Other models can be calibrated for relative naturalness of both function and structure, including a number of process simulation models that have been developed. Section 4 discusses models in more detail.

Several general issues have been raised regarding measures of naturalness. Most have to do with the representativeness of sampled attributes and how they ought to be weighed in any single measure of natural integrity. The measures used in models are typically gross, rather than specific, based on aggregate indicators of structure and function and usually limited to one group of organisms (e.g., fish, benthic invertebrates, birds), which may or may not be indicative of all ecosystem naturalness. Although ideally based on a thorough sampling of relative naturalness and humanly impacted conditions over the entire ecosystem (e.g., warmwater prairie streams in agriculturally modified areas), complete characterization of the variation among samples along a gradient is difficult to do inclusively for the range of human impacts that can occur. The meaning of relative integrity becomes more vague and difficult to interpret in complex settings altered in many interactive ways by human impacts.

Another issue has to do with sorting the effects of natural stress from the effects of human-caused stress. Many natural stressors produce the same effects as anthropogenic stressors. Fire, flooding, drought, and other stresses can be traced back at least partially to human actions as well as to natural causes. The response of ecosystems to natural and human-caused stresses is difficult to differentiate. In the same ecosystem context, natural ecosystem restoration occurs at the same rate. Measures of natural integrity following severe natural stress and severe human-caused stress can have indistinguishable results. Differences become more recognizable as the frequency, duration, intensity and pattern of stresses begin to change because of human impact. Thus measures of natural integrity following a single event in an isolated location, whether natural or not, is of questionable utility as are measurements made without knowledge of previous natural and human-caused events.

Related to this is the concept of natural succession. Locally stressed ecosystems "restore" naturally through a series of overlapping but different seral stages, each of which is natural. Each requires its own reference condition to establish natural integrity. Successional ecology is increasingly finding considerable variation in how succession proceeds and how it finally manifests in a more-or-less stable structure and function. Any number of natural states can result, some of which may be misinterpreted as human effect.

Another related issue derives from the importance of ecosystem scale of effect, both temporal and spatial, and how that importance translates into meaningful assessment of ecosystem condition. A full description of ecosystem integrity would include all of the defining historic conditions and resulting functions and structure over the entire ecosystem. For practical reasons, variations from natural integrity have been measured over relatively short time frames and a limited fraction of the entire ecosystem. Thus the representativeness of fully natural conditions and variations from them is sometimes questioned, especially with respect to long-term temporal variation. Because ecosystem functions associated with natural succession often act over decades and centuries, a temporally inclusive concept of natural integrity is difficult to develop. Because sites within ecosystems can naturally assume any of a variety of structural expressions (e.g., Holling 1973, 1996), the characterization of naturalness based on a few local reference conditions can artificially narrow the field of possibilities at any point along the gradient of naturalness.

The relationships between structure and function often are assumed to be close enough to use measures based on structure as an indicator of total ecosystem condition at the time of assessment. While the relationships of structure and function are becoming better known in general (as discussed later in this section), relationships in specific settings are typically more uncertain. A common indicator of structural component integrity is the biodiversity indicated by native species richness, which frequently correlates with ecosystem functional rate in simplified experimental communities (e.g., Tilman 1997) and in variety of field studies (Schlapfer and Schmid 1991). This relationship between function and structure is critical to understand for restoration purposes, and is discussed in more detail later in this section. More complex measures of integrity are multivariate including, in addition to taxonomic richness, other measures of taxonomic and functional composition, abundance and organism health (Karr 1991).

The concept of natural integrity alone offers no easy way to judge the relative merits of restoring naturalness among different ecosystems. Two or more types of ecosystems with very different structural and functional attributes can have the same index of integrity, indicating that each has the same fraction of remaining natural integrity. An ecosystem with full integrity composed of a few common species has as much natural integrity as a fully integrated ecosystem composed of many rare species. Similarly, a highly productive ecosystem may exhibit the same fully natural integrity as one of low productivity. Thus, the concept of natural integrity provides little insight into the *ecosystem services* or benefits linked to proposed changes in the structure and function of those different ecosystems. *It is a service-neutral concept*.

Inasmuch as ecosystem restoration seeks to augment natural services, an index of natural integrity can be a useful metric for evaluating restoration investment decisions if the relative completeness of ecosystem structure and function is highly correlated with the quality and quantity of services provided. Because service provision and relative integrity are not necessarily closely correlated, however, restoration plans guided by an index of natural integrity would not necessarily provide for the sustenance of species that are vulnerable to extinction (sensitive, threatened and endangered), or other services of significance. One rough indicator of potential service value is the relative scarcity of the more natural ecosystem condition at a national level. Scarcity of function and structure may indicate scarcity of associated services. Yet the species of commercial, recreational, vulnerable species support and other service relevance typically differ greatly in kind and abundance in different types of ecosystems, and even in the same type of ecosystem located in different geographical areas. Certain types of wetlands, for example, have been judged to be threatened and growing more scarce at a national level while they remain abundant (some would say overly abundant) in certain regions, such as Alaska. Thus, the national scarcity of specific structural and functional attributes is generally more critical for evaluating and justifying ecosystem restoration projects.

#### 3.2.2.2 Ecosystem Integrity, Sustainability and Scale

Odum (1993) suggested that the functional capacity of ecosystems to *sustain* diverse human services is the most fundamental natural resource requiring management stewardship. Diverse interpretations of the concept of ecosystem *sustainability* are encountered in policy such as that of the U. S. Forest Service management goal (Federal Register 2000) and national goals associated with economic development (e.g., The Presidents Council on Sustainable Development 1996; NRC 1999b). Virtually all of these concepts either explicitly or implicitly link the sustainability of ecosystem function and structure to the reliability of natural resources and natural services.

At least two important concepts of ecosystem sustainability can be identified among such goals. Ecosystem sustainability is the maintenance of all natural parts and processes necessary for maintaining ecosystem integrity through a self-restorative process following local ecosystem disturbance. The *g*enetic information stored in species falls into this category because it provides the design guidance for restoring many of the natural parts and functions of ecosystems. An associated concept links the conservation of ecosystem functions with the capacity of ecosystems to accommodate environmental stress by transforming adaptively to other *self-regulating* states (Hollings 1973 and 1996). The variety of self-regulating adaptive states and the capacity to adapt are maintained as long as the genetic information controlling the process remains extant and accessible in species living within the ecosystem. The stress may be natural or, if human-caused, may be intentional (managed) or unintentional.

Ecosystem sustainability typically is described in terms of temporal dynamics, but is greatly influenced by the spatial scale of the dynamics and the pattern of the natural ecosystem expressions remaining during and following stressful disturbance. Local integrity in small fractions of ecosystems often varies naturally from ecosystem-wide integrity. Such local alterations occur naturally through climatic, disease and other natural stress, and are restored naturally through residual capacity for self-repair and, very importantly, through recolonization from unimpaired source areas. Natural loss and recovery of local integrity happens "routinely" when floods, fire and other extremes decimate only small portions of ecosystems. Immediately following a local flood event, for example, the species richness and integrity may be temporarily decimated while the remaining watershed system of similar streams changes little. The rate of recovery after stress removal usually increases as the intensity and size of the impacted area decrease and as the boundary between disturbed and undisturbed areas becomes more irregular.

Orientation and location of the locally disturbed fraction within the larger ecosystem also are important determinants of natural restoration rate and completeness. Especially influential are connecting vectors of wind, water, and other transport processes and the conditions of natural features connecting different ecosystem fragments. Disturbances at the edges of ecosystems tend to be less certain of full recovery and more likely to transform to adjacent ecosystem attributes than disturbances toward the centers of ecosystems. Even less certain is restoration of small and isolated patches of ecosystem far removed from other natural vestiges of ecosystems that have been largely converted to other structures and functions.

Natural integrity is permanently degraded once unique parts and processes are permanently lost, such as by species extinction. Otherwise, natural integrity is only locally altered to another state until that time when the stresses naturally wane or are eliminated through management and the naturally restorative process can proceed. Except for the intensity and duration of stress, which are typically increased by human action, many physical forms of human impact are difficult to differentiate from natural stresses (e.g., accidental fire, logging, flooding, lake formation, levee development, fire, invasion by new species). Other human impacts are globally pervasive, often systemic and more persistent, such as some chemical and climatic alterations. These are the most troublesome because they do not respond to localized restoration actions and may sometimes limit the effectiveness of restoring the most desirable ecosystem function and structure.

Human-caused stresses (e.g., dams, stream dredging and pollutants) also have locally transforming impacts, which, even after many years, can recover quickly to full natural integrity once the stresses are removed   as long as enough natural ecosystem remains intact and well connected to the restored site. Certain stresses are more difficult to remove than others, however, such as refractory chemical or radiological contamination. Physical stresses typically can be eliminated more quickly. The potential rate of natural ecosystem restoration decreases as more of the natural structure and function is replaced with artificial features, function, and maintenance.

At some point, the combination of human and natural stresses accumulates enough to overwhelm natural recovery and the ecosystem-wide integrity and sustainability declines as unique parts and processes permanently disappear. While natural evolution of new genetic information tends to balance natural loss, exceptional human impact results in a net loss as extinction exceeds generation of new genetic information. This attrition of parts and processes limits the array of possible manifestations of ecosystem structure and function. From a management standpoint it becomes increasingly costly as it increasingly limits management choices.

Restoring the natural connections of degraded ecosystem areas to the largest remaining patches of natural ecosystem structure and function is an important key to management success in recovering threatened parts and processes to a sustainable state. Even when the past service conditions of degraded areas adjacent to natural areas with desirable natural services are less well documented than service conditions at sites far removed from the remaining natural ecosystem, the risks of recovering the desirable levels of natural services are likely to be lower at the adjacent sites where system connections are complete.

Whether ecosystem integrity or sustainability should be targeted for protection and restoration. Some natural resource managers prefer to emphasize ecosystem sustainability over ecosystem integrity because they believe integrity is less readily measured and evaluated than is sustainability (e.g., Link 2002). This preference depends somewhat on whether structure or function is more important to the manager. It also seems true that integrity is most often linked to structural attributes and sustainability is more likely to be linked to functional attributes, such as production. Link (2002) for example, noted that while ecosystem structure often changes locally those local areas of "ecosystems will continue to function, albeit at different configurations" of structure. An ecosystem area under the "stress" of resource use and management can result in a range of sustainable functional states depending on management objectives and system manageability. Typically, the structure of these different functional states are dominated quite predictably by relatively common plant and animal species. Sustaining specific compositions of scarce species in such locally managed area of ecosystems proves to be a more difficult thing to do, however. Reliable maintenance of rare species typically requires a larger scale of management consideration, including natural areas set aside from management. Despite the apparent differences, the concepts of integrity and sustainability are closely related and similarly depend on the spatial extent and patterns of ecosystem alteration by human activity.

The nearly universal manifestation of human impact among ecosystems may make measures of ecosystem sustainability more practical criteria for characterizing natural integrity than natural reference conditions. Fully natural conditions are increasingly difficult to find in many ecosystems. However, the choice of functional and structural indicators for judging sustainability is critical. If the emphasis is on sustaining all structure and function for future management options, sustainability ought to be gauged by the condition of the most vulnerable of irreplaceable parts and associated processes. Once extinct, these parts and processes compose the lost integrity of ecosystems.

The threat of permanent loss of ecosystem parts and processes often can be thwarted by management, but only if the status parts, processes and threatening conditions is tracked and conditions are restored at least to the minimum of ecosystem naturalness needed to assure sustainability. Examples of such tracking is the database, “NatureServe”, which is maintained for state Natural Heritage programs and other users, and the listings of species status in the Endangered Species Act. These lists of vulnerable parts are among the clearest indicators of threatened natural integrity and sustainability of ecosystem attributes.

### 3.2.2.3 Ecosystem Integrity and Biodiversity

Some indicator of *native biodiversity* is the typical measure of ecosystem integrity. While native species richness is a common indicator of biodiversity, and sometimes is assumed to be synonymous, current concepts of biodiversity hold that it is more complex and comprehensive than species richness alone. This multidimensionality and comprehensiveness is revealed in the recent definition of Redford and Richter(1999): “*Biodiversity refers to the natural variety and variability among living organisms, the ecological complexes in which they naturally occur, and the ways in which they interact with each other and with the physical environment”*. The definition used in Heywood (1995) adds nuance to this inclusive definition:

> “...biodiversity is defined as the total diversity and variability of living things and of the systems of which they are a part. This covers the range of variation in and variability among systems and organisms, at the bioregional, landscape, ecosystem and habitat levels, at the various organismal levels down to species, populations and individuals, and at the level of the population and genes. It also covers the complex sets of structural and functional relationships within and between these different levels of organization, including human action, and their origins and evolution in space and time.”

All variation and variability in ecosystem the structure and function determined by life form and process is included in this comprehensive concept of biodiversity, which provides a theoretically complete measure for natural ecosystem integrity *and more*. Human alterations are also included in this broad definition, which is more consistent with the ecological concept of naturalness than with the social concept. Such comprehensive definitions of ecosystem biodiversity closely approach definition of all of the structure, function, and other processes composing ecosystem integrity, including human impact. But importantly, *biodiversity is more meaningful at a national level of ecosystem differentiation* because different ecosystems of the same integrity always have different expressions of biodiversity.

As inclusive as biodiversity is, it does not include those physical attributes of the ecosystem environment that are not a product of life processes. Where one ends and the other begins is difficult to determine, however. But the most physical of forces and constraints that fundamentally shape and drive ecosystems contribute to a larger ecosystem diversity, more inclusive than biodiversity. Such basic properties include the light entering the ecosystem, gravity, strong and weak forces in matter, the geological foundation, much of the topography, much of the hydrology, and some of the climatology. These are the physical inputs that are most fundamentally restored, if altered, to reestablish the natural ecosystem. Hydrology and topography are most emphasized by Corps restoration policy. Even these are influenced by life processes (through watershed and atmospheric processes) to an extent that may be difficult to assess, but is necessary for accurate forecasts of ecosystem response to management actions. Relevant to Corps restoration policy, biodiversity is an inclusive measure of those ecological resources that are a function of habitat restoration and the basis of gauging ecosystem restoration effectiveness.

When comparing ecosystems, biodiversity is a better indicator of self-regulating function, functional stability and sustainability of attributes than is natural integrity. Some ecosystems have lower biodiversity and functional stability than others of equal natural integrity. Ecosystems of low natural integrity are particularly vulnerable to great change when a new species invades them. A good example of the great change that can come about is the transformation undergone by lamprey and zebra mussel invasion in the Great Lakes ecosystems, which had relatively low natural biodiversity. Whether or not the lakes are becoming more functionally stable is yet to be determined, but probably depends on the extent to which native species are totally extirpated by the new species.

Biogeography in general reveals that many, if not most, species naturally invaded ecosystems in the past. Recent northward extensions of some species (e.g. the Virginia opossum and Cardinal) into different ecosystems is an example of a natural invasive process that is not necessarily destabilizing. Many past species invasions may have added to ecosystem biodiversity more or less immediately upon entry into the system while others may have decreased biodiversity, at least in the short run, by driving other species to extinction. However, Mora et al. (2003) maintain that the local fish species richness in many reef ecosystems is sustained by dispersal from biodiversity “hotspots” in the Indian and Pacific Oceans. This suggests a net stabilizing influence of natural invasion.

A number of species have been introduced in the U. S. for their recreational and commercial value, for example, without clear negative impact on global biodiversity (e.g. brown trout and ring-neck pheasant). Some of these species seem to have partially replaced native species contribution to ecosystem structure and function while increasing biodiversity and ecosystem service value. Also of interest is whether the means of invasion—by human vector or other means—makes a fundamental difference. This area of scientific questioning has definite implications for the concept of ecosystem naturalness and its measurement. Because biodiversity, sustainability of function and structure, and naturalness are often assumed to be closely correlated, these differences have restoration implications where the stated purpose is greater naturalness, as it is in Corps policy.

A simple species richness measure of either biodiversity or integrity is limited in scope and can miss important aspects of biodiversity, especially the physical variation in habitat, depending on how closely species richness correlates with other ecosystem variation and variability. Inclusion of habitat —the physical part of the ecosystem complex— and landscape —a mix of physical and biological elements— extends the concept of biodiversity well beyond a community measure and toward a more complete measure of ecosystem diversity most relevant for ecosystem restoration plan formulation and evaluation. Based on the definitions of Redford and Richter (1999) and Heywood (1995), virtually any change in the abiotic and biotic structure and function of a natural ecosystem would result in a biodiversity change, which might be measured to document when the variation inherent in an ecosystems natural integrity is surpassed and integrity is lost.

No existing quantitative measure of biodiversity has approached the system-wide inclusiveness of the Redford and Richter (1999) and the Heywood (1995) definitions, however. Most quantitative measures are indices based on the relationships existing between a relatively small selection of habitat attributes and certain biological attributes of ecosystems. The quantification may be as elemental as a species-area relationship (see Rosenzweig 1995), which generally demonstrates the relationship between the ecosystem area sampled and the number of species encountered. Numerous other ecosystem indices have been developed to include more community and habitat attributes and relationships, but relatively few have been applied beyond the site for which they were developed.

Where ecosystem feedback processes between community and habitat are essential to ecosystem characterization for plan formulation and evaluation, models that simulate the ecosystem state dynamics and incorporate the feedbacks may provide more insight for informed decision making. Dynamic state, process simulation models (e.g., DeAngelis et al. 1989, Bartell et al. 1999) also can be regarded as simulations of ecosystem biodiversity as defined broadly in Heywood (1995). In contrast to indices, they usually estimate how actual output concentrations, numbers, and other measures of populations, communities and abiotic processes might change, given changes in model-input conditions. But because of the complexity of real ecosystems, process simulation models, like all other models, cannot completely represent the diversification process in ecosystems and must rely on the capacity of aggregate biodiversity measures (such as functional guilds of species) to indicate habitat suitability for all species.

### 3.2.2.4 Ecosystem Integrity, Cultural Integrity, and Health

One difficulty encountered in the definition of natural ecosystem integrity is the growing scarcity of fully natural states. Studies in the most remote ecosystems indicate that few ecosystems are free of human influence and all within reach of civil works management have undergone some cultural modification. Numerous ecologists accept the impracticality of either protecting or restoring most ecosystems to fully natural states. They emphasize reference to a more natural state rather than to a fully natural state. Regier (1993) concluded that a practical “notion of ecosystem integrity is rooted in certain ecological concepts combined with certain sets of human values” resulting in a state of *cultural integrity*. Humanly modified ecosystems exhibit cultural integrity when they sustain a satisfying combination of natural services and artificially enhanced services both locally and globally. The idea that humanity can and should integrate smoothly into the natural workings of ecosystems is fundamental in the philosophy of environmental sustainability.

The Society For Ecological Restoration, for example, has defined restoration in terms of recovering and managing ecological integrity, including "sustainable cultural practices". Just as for natural integrity, the restoration of ecosystem integrity that includes sustainable cultural practices, or cultural integrity, would normally refer to historic description or existing reference conditions. Thus the concept of ecosystem cultural integrity is linked closely to the concept of human-welfare and environmental *sustainability* ("people on earth...meet their needs while nurturing and restoring the planet's life support systems"—NRC 1999b) and can have as much or more policy meaning with respect to cultural practice and resulting services as ecological meaning.

The concept of cultural integrity has much in common with the concept of *ecosystem health* (Costanza 1992). Both take much of their meaning from the sustainability of function and structure and the desirability of associated natural services. *A healthy ecosystem is one that is both sustainable and culturally desired*. Healthy states of cultural modification often are preferred over more natural states based on threats and opportunities associated with human health, property, and other sense of human prosperity. Therefore, a more natural ecosystem justifies restoration at a diminished site only when the perceived value of the restored services exceeds the benefits eliminated by the restoration.

Achieving cultural integrity and ecosystem health requires determination of the minimum spatial and other resource requirements preserving all relatively natural states in an ecosystem as it becomes culturally modified. In some cases, restoration will be required to assure continuity of relatively natural states. At some point, any further conversion of natural ecosystem process will threaten to compromise cultural integrity and ecosystem health. In those cases, commitment to maintenance of cultural integrity will require either cessation of cultural transformation or restoration of some parts and processes of natural ecosystems before other parts are culturally transformed. The underlying assumption is that functional and structural sustainability is possible in various states of cultural integrity as long as all of the ecosystem parts and processes remain available to convert to other ecosystem states when management objectives change. The *first rule for sustaining restoration options* is to maintain some minimum inventory of ecosystem parts, starting with species.

The most usual strategy used to restore cultural integrity and health *locally* is to improve habitat quality and connectivity within the physical limits of beneficial enhancements, such as improving clean water in modified waterways and harbors. This is most often achieved through structural engineering. In contrast, deficiencies in ecosystem integrity and health can be approached *globally* by restoring and sustaining the viability of all parts and processes contributing to natural integrity. This is most often achieved by removing the effects of past structural engineering.

### 3.2.2.5 Challenges to Managing For Natural Integrity

In principle, nothing short of maintaining a substantial fraction of ecosystems in a relatively natural state is necessary to assure that all parts and processes will be sustained. However, the precise fraction needed and its landscape position and integration usually are poorly defined and the methods for doing so continue to evolve rapidly. Restoring and protecting natural fragments of ecosystems large enough to sustain future management choice may require larger areas than first anticipated and active disconnection of the remaining native ecosystem from contaminated parts of the ecosystem. Changes that pervade and permeate throughout an ecosystem are among the major impediments to sustaining the fully natural state and are common threats to sustaining native biodiversity.

Potentially degrading pervasive changes include widely dispersed contaminants, global climate change, invasive nonnative species, and complete conversion of ecosystems to other physical forms. Contaminant removal can take decades following discontinued use of ecosystems for waste reception. Similar delayed responses can be expected for ecosystems that have undergone extensive physical transformation, such as changes in the flow, sediment loads and temperatures of natural river systems. While natural ecosystems can adapt to climate change along elevation and latitude gradients, the prerequisite space needs to be available. As has been mentioned, aggressively invasive species allowed access to ecosystems by human activity can cause large and permanent changes in the composition of ecosystems and significant changes in ecosystem function. Removal of invasive species often proves impractical once they become well established.

Thus, the most desirable combination of ecosystem services possible in ecosystem settings will rarely if ever occur in a fully natural ecosystem, even when the sole objective of management indicated by public consensus is to preserve or restore the most natural ecosystem possible for whatever array of services will result. The more practical restoration investment questions focus on determining how the extent recovery of a *more natural* ecosystem condition results in a more socially desirable mix of natural and enhanced services

### 3.2.3 Integrating Enhanced and Natural Services

### 3.2.3.1 In Search of the Ideal Result

The concept of sustainable development implies continued improvement of the human condition though seamless integration of natural and artificially enhanced resources for optimum delivery of services. A relevant project-planning question asks: What is the proper emphasis placed on artificial enhancement of certain ecosystem resources and provision for natural ecosystem resources to sustain the most beneficial combination of services? With respect to water-resource management, ecosystem restoration appears to be justified in those instances where previous artificial alteration of services has replaced natural services of greater social value.

Few would argue that artificial enhancement of specific ecosystem services entailing some *local* loss of ecosystem naturalness has not resulted in improved public welfare. Principles of water, forest, range, farmland, recreation-land, urban-land, fish, and wildlife management are based on the assumption that at least some artificial alteration of natural service is beneficial in the proper context, even when native biodiversity is locally depressed.

Few would argue, on the other hand, that what was thought to be enhancement of certain resources and services in fact ended up diminishing total benefit by unintentionally eliminating too much beneficial natural service. Correcting for greater benefit is merely costly when all of the parts and processes can be restored, but possible when justified based on the perceived benefits and costs. When parts and processes are entirely lost, complete restoration is impossible and no amount of corrective management can replace the lost management options

The Corps has received its environmental-improvement authority from Congress in recognition of correctable service deficiency and resource degradation.   Much of the environmental legislation of recent decades is intended to reverse degradation of the general public welfare as a consequence of less than optimal management of natural resources and their use. Achieving this result requires identifying the desired mix of natural and artificially altered services and linking them back to the underlying resources.

### 3.2.3.2 Identifying The Desired Mix of Services

Artificial alteration of natural resources and services continues to enhance social benefits until the integrity of the natural system is so compromised that the sum of natural and enhanced service benefit begins to decrease (Figure 3.1).  A general example is the accumulation of water-control structures for enhancing navigation and flood-damage reduction services that has contributed inadvertently to the growing scarcity of globally unique organisms.  The exact relationships between natural and culturally enhanced services and their combined benefits vary from ecosystem to ecosystem and from one social context to another.

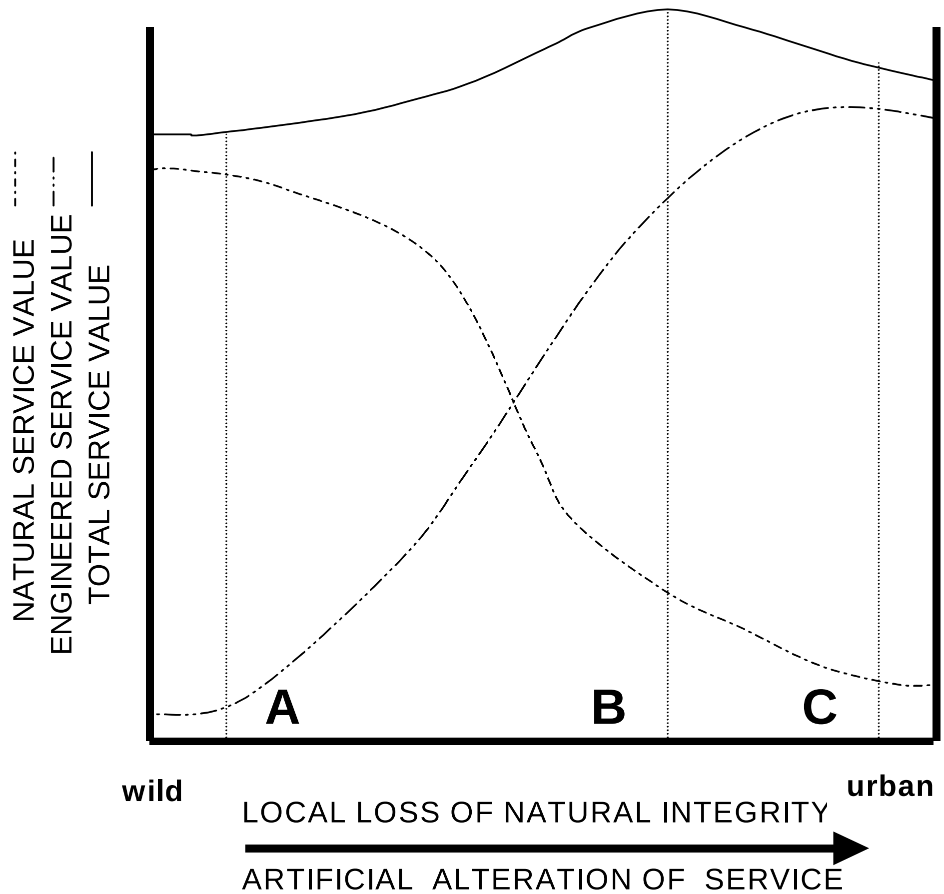


**Figure 3.1. A conceptual example of value changes associated with natural service benefits and artificially enhanced service benefits along a spatial or temporal gradient from wild to highly urbanized condition. In this concept, all environmental and economic costs and gross benefits are assumed to be additive using some common unit of measure. At Point A on the gradient, the ecosystem is quite wild and most valued for its natural services. At point B, the combine natural and artificially enhanced service benefits are maximized . At point C, the ecosystem is artificially altered (urban development) to a point where most value is from development that has gone too far toward displacing natural services, resulting in decreased total benefit.**

To illustrate the general point, three different ecosystem conditions are conceptually represented in Figure 3.1 at points A, B, and C along a gradient from fully natural ecosystem integrity through increasing degrees of cultural modification. As described by Regier (1993), each of these three states has come into an ecological equilibrium that sustains “an organizing, self-correcting capability to recover toward an end state that is normal...for that system.” even though specific conditions at points B and C vary greatly from the most natural conditions at point A. These are all, therefore, sustainable states. At point B, the ecosystem is providing close to the mix of natural and enhanced services that provide maximum public benefit. At point C, the alteration of natural services has gone too far to provide the maximum benefit.

Transforming the concept presented in Figure 3.1 into practice is complicated because different services are not equally amenable to monetary valuation and summing monetary and non-monetary measures of benefit and cost has to be subjective. The inability to readily estimate economic values for certain natural services probably has contributed to a contemporary public sense that past water resources management has inadvertently degraded significant natural resources enough to warrant their recovery through restoration measures.

The Corps often is involved with some intermediate ecosystem condition broadly bracketing point B in Figure 3.1 where a more even mix of natural and artificially enhanced services are provided and where the combination of services at least in theory approaches maximum public benefit. These intermediate ecosystem conditions frequently have a more balanced mix of natural and artificially enhanced services than either the wilderness or the urban extremes, but not necessarily the optimum mix for maximum benefit. Further enhancement may be justified when the mix overemphasizes naturalness for the social wants and needs. Restoration may be justified when the mix overemphasizes enhanced services. A condition of overly enhanced services can result either because of past mistakes in judging the proper mix for maximum benefit, or because societal preferences have changed.

The past water resource engineering done to enhance services defined by authorized purposes (e.g., navigation, flood damage reduction, water supply, recreation) had to be economically valued in order to justify its construction in the first place. The preference for those enhanced services may have changed from the past, but can still be valued using the same techniques that justified the enhancement in the first place, as defined in Federal water resource management Policy (WRC 1983). At point C, where traditional water resources enhancement measures now dominate all service provision, most of the service has been and could now be economically valued according to national economic development criteria.

Where little service enhancement has occurred , as at A in Figure 3.1, much more of the ecosystem service is likely to be environmental than economic. However, some of the natural ecosystem output might be valued for its service much as it was for enhanced states. For example, natural rivers have navigation properties that can be valued just as culturally modified waterways are valued. Similarly the recreational service of natural rivers has been economically valued much as the recreation of reservoirs and waterways have been valued to evaluate their development. Some of the natural services of wetlands also have been monetarily valued as well (see Heimlich et al. 1998 for a review), albeit at different levels of confidence depending on the service and knowledge of natural function and structure. Other natural services have not been so confidently valued, such as the value of restoring natural ecosystem support of species vulnerable to extinction.

### 3.3 Ecosystem Outputs: Natural Resources In Support of Services

With respect to restoration *plan evaluation*, the ecosystem outputs of particular interest are the *significant natural resources* that both directly and indirectly underlie ecosystem services. The natural resources of concern in the Corps environmental policy have ecological attributes (WRC 1983). The concept of natural resources is as much social as it is ecological, being the "store" of materials and potential energy of immediate or possible use to humanity. Resources with ecological attributes fall into the general subcategory known as renewable resources, which are regenerated through *life processes.* In addition to living resources affected by life processes, renewable resources include numerous non-living products, such as the dead organic portion of soil and the water discharged from watershed influenced by life processes.

Traditional concepts of renewable natural resources focus on extractable resources such as the resources harvested in commercial fishing, duck hunting, timber, and livestock forage consumption. More contemporary concepts include nonconsumptive use, such as recreation based on observing nature or setting aside habitat use for endangered species. Underlying these resources, whether extracted or not, is a complex interactive network of nonrenewable and renewable structures and functions that provide for all of the energy and material needs of the used resources, including such basics as light, inorganic sediments and solutes, and water. This interactive complex of underlying natural resources comprise ecosystems that are indispensable for renewing the resources of direct utility and thereby take on significance and value indirectly through the used resources. While restoring a more natural state may include what is necessary to restore the meaningful, significant resources, it also may not when restoration is partial or if the history of the significant resources and the most natural condition is unclear.

This subsection summarizes some of the more important ecological concepts about how ecosystems work indirectly through function and structure to renew and sustain natural resources providing goods and services to humanity. Any attempt to rank the importance of the ecosystem outputs reflecting the functions, structure and sustainability of ecosystems immediately recalls the proverbial chicken and egg. Teasing apart ecosystem process risks loosing concept much as the forest gets lost from view as we focus on the trees and other parts. The order of discussion for topics in this section implies less about relative importance of ecosystem properties than it does the relative complexity and uncertainty of the principles. The four categories of structure and function used to develop this discussion start with energy flow and biomass production, followed by material flow and cycling, and hydrologic process, and culminating with self-regulation, functional stability and ecological sustainability. Each of the four categories of function and structure are compared to biodiversity measures for their potential utility as measures of ecosystem output for Corps restoration planning purposes.

### 3.3.1 Production, Biomass and Other Energy-Flow Outputs

#### 3.3.1.1 The Relevance of Energy in Natural Structure and Function

All management of biological resources depends on natural *energy-process* dynamics to sustain renewable resource function and structure through the production and maintenance of community biomass (Figure 3.2). No other ecological output reveals more possibility for universal application than potential and kinetic energy in ecosystems. Energy is the one universally distributed natural resource found in all ecosystem form and process that can be compared as Joules or other unit of energy. However, like naturalness, *it provides little insight, beyond power supply*, into natural or management-enhanced *ecosystem service values.* Society examines relatively few services and their tradeoffs in terms of net energy gain and loss and, although a related concept of power maximization has been used to explore societal decision process (Odum 1971), the concept remains obscure and peripheral.

Because the diversity of natural resources associated with ecosystems appears to be important in determining the total service value of ecosystems, the processes by which community-level production is distributed among species groups, individual species and other resources are of great interest (Figure 3.2). While some natural resources are produced in large community aggregates of numerous species, such as the capacity of vegetation to store carbon and regulate greenhouse gas accumulation, most natural resources are uniquely linked to services provided by a small fraction of the ecosystem's species. For example, forests produce wood resources with a wide spectrum of uses, each tree species in the forest producing wood with a unique service quality. The raw energy value for fuel is only one source of value. Vertebrates provide recreation, but various species provide unique opportunities for recreation with different service values. Even endangered species are not treated as if they have identical value, some getting more protection investment than others. The diversity of services provided by an ecosystem contributes to how much they get used and how highly valued they are. That service diversity and value depends on the extent that production and biomass are partitioned into recognizably different species and other ecosystem forms and functions.

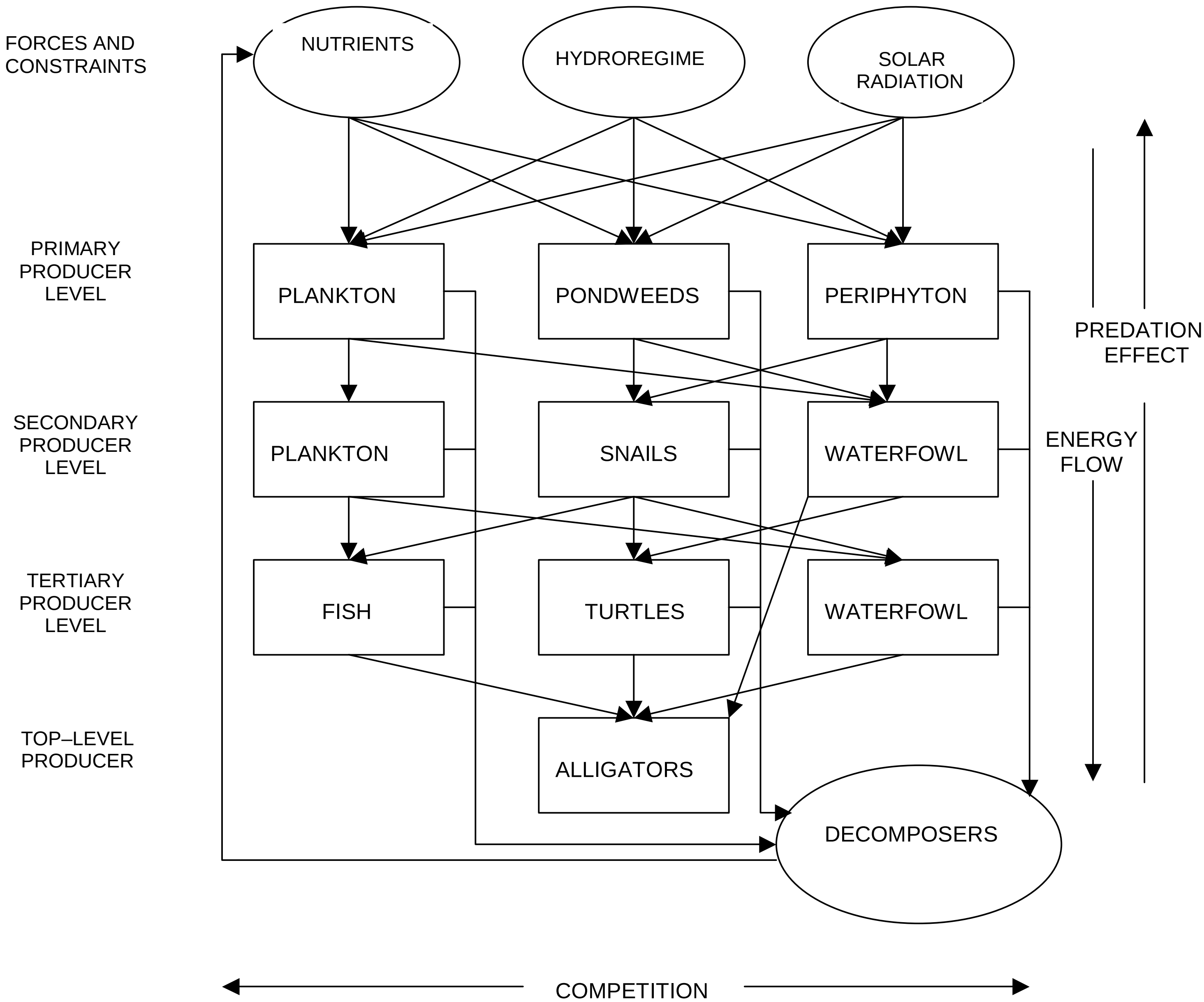


**Figure 3.2. Community partitioning of energy and nutrient materials is determined by energy loss at each feeding (trophic) level, partitioning of resources among species in each trophic levels, and predation effects between trophic levels. Each level is occupied by many more species than indicated here by a few functional groups. In riverine and coastal systems environmental hydrodynamics influence resource partitioning at all points in the system**.

### 3.3.1.2 Energy Transformation and Partitioning Into Resource Production

A basic natural limitation to the development of ecosystem biomass (potential energy) and biodiversity (diversity of potential energy forms) is the availability of energy that can be transformed into living process and transmitted from one living form to another through food webs. Any transformation of energy from one form to another via primary production and dependent food webs is accompanied by energy output in the form of heat. Ecological process is quite inefficient at transferring life-generating energy from one consumer trophic level to another. A large amount of energy is lost from the food web as heat with each transformation of energy from photosynthesis through subsequent food-web transformations. For this reason, food chains are of limited observable length, typically revealing only 4-5 energy transformations from primary producers to top carnivores (Pimm 1982).

### 3.3.1.3 Biodiversity and Production Relationships

The relationship between biodiversity, organic production, and service delivery is determined in part by the amount and reliability of energy available to primary producers, the energy loss in transformation, and the amount of resource partitioning among different life forms within each of the production levels. The complexity of resource partitioning, as indicated by species diversity, influences community persistence in the presence of destabilizing events and functional resilience following disturbance. Empirical tests support this theory more often than not (Tilman and Downing 1994; Naeem et al. 1994; Tilman 1997; Naeem 1998; Walker 1992, 1995, Schlapfer and Schmid 1991). Modeled relationships (e.g., Figure 3.3) are being refined and general concepts are gaining wider acceptance as they are tested with empirical data from other situations.

The insurance analogy, for example, is offered to explain how biodiversity increases stability of community production (Shigeo and Loreau 1999). Compared to a community with few species, a diverse community has a larger selection of adaptations to draw from as environment changes. While the relative production of species changes with changing environments, the total community production is sustained.

Tilman (1997) has provided both hypothetical models and empirical tests that indicate biodiversity is related to production and to resilience as measured by the recovery of productivity following drought disturbance. If native species richness proves to be a consistent predictor of functional performance in various ecosystems, it may serve as an index to natural service provision, such as sustained reproduction of genetic information and the reliability of an array of natural services aligned with ecosystem functions. Enough research has been done to recognize that numerous exceptions occur, although it is not always clear why. Schlapfer and Schmid (1999), in a comprehensive review of studies, show that positive relationships such as Tilman's (1997) occur much more often than negative relationships, but numerous other studies reveal no relationship. More research is needed to determine why the exceptions occur. Relatively little study has been conducted in freshwater.

The relationship shown in Figure 3.3 also indicates that, at any one time, *a small fraction of the species contribute to a large fraction of the productivity and to the resilience* as measured by return to previous productivity level. However, as climatic, hydrologic and other conditions in the ecosystem vary naturally the contribution of each species to total ecosystem function also shifts.  Some common species become less common and some

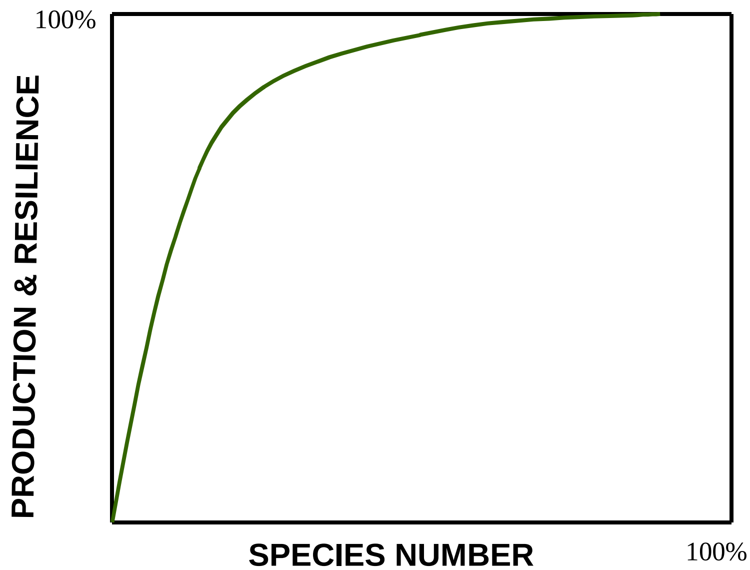


**Figure 3.3.  General relationships between a species richness measure of biodiversity and production and resilience functions in simple and complex systems (Based on information presented by Tilman 1997).   Most ecosystem function is associated with common species. Most species ordinarily contribute much less to production-related functions.**

rare species become more common until conditions change again.  In this way, diversity sustains higher total production and more stable production and biomass.  Predictable patterns of environmental variation maintain suitable conditions for all species some of the time.  Each species has evolved means to persist through stressful periods. Nonnative, invasive species can play an important role because they are most likely to dominate production in a disturbed system and greatly displace the original diversity while restoring and sustaining production.

### 3.3.1.4 Relationships Between Biomass and Physical Process

Feedbacks between biomass generation and physical processes are common in ecosystems and cause physical inputs to change into significantly different output attributes, which often serve as the inputs to other ecosystems.   For example, as biomass accumulates in watersheds the hydroregime outputs typically become more stable and substantially alter the physical input of water flow and amount into aquatic ecosystems. Aquatic species adapted to that hydroregime disappear quickly once the watershed is substantially disturbed by natural or cultural processes and the hydroregime becomes less stable.  Another example of functional feedback control by accumulating biomass is the self-shading caused as photosynthetic biomass accumulates, limiting the rate of photosynthesis. Light is greatly reduced by terrestrial vegetation overhanging aquatic ecosystems or by algae within aquatic ecosystems, greatly reducing light input, altering light quality and influencing the amount and type of productivity in the underlying ecosystems.

#### 3.3.1.5 Which Is The Better Measure of Integrity—Production or Biodiversity?

Is community production or biomass a more appropriate measure than biodiversity for formulating for ecosystem integrity and evaluating its restoration? As suggested by Figure 3.3, much community production and biomass is commonly associated with a small fraction of the species, some of which may be nonnative invasive species. Productive and abundant species often are less sensitive to habitat attributes than are those rare species that are valued most for their endangered status. The most productive species may not be indicative of the habitat needs of all community members, including the endangered species. Restoring community-level production based on the habitat needs of a few dominant species will not necessarily result in restoration of the rare species. In contrast, restoring ecosystem conditions, including removal of nonnative species, that will reestablish most of the native species richness, is more likely to restore community production close to the level of natural integrity. This is likely because the community members evolved with a set of native habitat and community features that determines the total production and it's partitioning among species.

### 3.3.2 Material Flow and Cycling Outputs

#### 3.3.2.1 Material Types and Relevance

A close and necessary association exists among material flows, energy flows, production, consumption, and decomposition as schematically illustrated in Figure 3.2. Energy flow and resource partitioning drive material cycles and other material flows that have measurable ecosystem outputs in the form of material concentrations, densities, loads, transport rates, and dispersal rates. Materials include all matter organized into life forms, geological foundations, atmosphere, soils, and sediments and all matter in transport suspensions and solutions. Among the most prominent are nutrients required for life structure and function, inhibitory toxic materials, habitat substrate materials, and transport materials. Many ecologically active materials are transported by water, wind and other physical-transport materials connecting within and among ecosystems. Other important material-flows manifest in organism dispersal. How resource materials are distributed among diverse populations of living organisms determines the production and biomass of each population and the resulting biodiversity.

#### 3.3.2.2 Nutrient Limitation and Materials Cycling

Where liquid water is plentiful, nutrients such as nitrogen, phosphorus and iron are commonly in least supply for the production demand and limit production rates. When habitat changes increase or decrease supplies of limiting nutrients, marked changes in community production typically result, accompanied by changes in ecosystem structure and function. Nutrients are cycled through ecosystems (Figure 3.2) as they are taken up in primary production and broken down by consumption and decomposition.

For nutrients with gaseous pathways—including carbon, hydrogen, oxygen and nitrogen— the scale of cycles typically are global in scope. Carbon dioxide released from one ecosystem conceivably can be taken up by ecosystems thousands of miles away. Sedimentary nutrients, such as phosphorus and iron, have no atmospheric pool and are typically cycled at smaller scales than nutrients having a gaseous cycle. They are more likely to be the nutrients limiting growth and production, and more likely to be cycled "tightly" within ecosystems, conserving them for future use.

Cycling efficiency is related to interactions between biomass elaboration and erosion forces operating in an ecosystem. Where erosion and transport forces are moderated by biomass type, amount, and distribution, as in a well-rooted forest or grassland, recycling efficiency is high. Where net erosion and transport forces are high, as on a well-watered steep gradient without much rooted biomass, recycling efficiency is low. Recycling efficiency is closely associated with the amount and distribution of biomass maintained in a system, which also is associated with biodiversity and productivity. Tilman (1997) discusses how, up to a point of greatly diminished returns, adding diversity to primary producers increases total productivity and the efficiency of nutrient uptake in plant tissues**.**

### 3.3.2.3 Ecosystem Boundaries, Interactions and Material Flows

Ecosystem boundaries are most typically defined by interruptions or abrupt changes in material flow rates and form. Watershed boundaries are among the most obvious for freshwater communities, but other topographic features and vegetation-defined boundaries most influence terrestrial communities. Yet, even in the most developed and clearly bounded ecosystems, material retention is rarely 100 percent. Ecosystem boundaries are naturally porous and the structure and functions of some ecosystems have evolved to depend on a reliable supply of materials "exported" from other ecosystems.

Thus, the rates, fluctuation, and extent of material flows among ecosystems greatly determine the interactions between habitat and communities within and across ecosystems. The nutritional outputs from one ecosystem usually are inputs required to sustain function in other ecosystems; thus, most ecosystems are linked closely with other ecosystems. Streams and lakes would be lifeless without wind- and gravity-driven loss of nutrients from watersheds, channels and basins. Similarly, deep lake and ocean bottoms would be lifeless if pelagic ecosystems were 100% efficient in recycling nutrients.

Because species diversity typically increases as the physical diversity of habitat increases (e.g. Rosenzweig 1995), the inefficiency of nutrient cycling probably has resulted in greater global biodiversity than if the ecosystems at the top end of gravity gradients were 100% efficient at retaining nutrients. Too much of a good thing also is problematic. The increased nutrient, sediment, and toxic-material loading resulting from watershed disturbances (e.g., crop culture, grazing) remains among the most pervasive of water quality problems complicating aquatic ecosystem restoration.

Even in the least disturbed states, small depressional aquatic ecosystems are naturally ephemeral, filling quickly with the materials exported from watersheds. Unique species diversity typically occurs in the oldest and largest lakes. Those communities adapted to small lakes and wetlands typically depend on creation of new habitats as old ones are eliminated.

#### 3.3.2.4 Which Is The Better Integrity Measure—Material Flow or Diversity?

Is material flow and cycling associated with natural ecosystem integrity a more appropriate target for restoration than biodiversity? Determining what would serve as appropriate indicators of complex material flow processes is the first need. Biomass, production, and decomposition together provide a crude indication of the amount of material that might be taken up by, stored in, and released from an ecosystem. Because ecosystem biomass, like production, can be restored to a close approximation of original ecosystem biomass in relatively few species, either natural or exotic, the habitat and community needs of rare species containing unique genetic information could be easily overlooked. Less is known about decomposition, but some evidence suggests that a few dominant species can dominate this process as well. Past use of exotic plants to "restore" eroding banks and watershed often restored erosion rates to a close approximation of natural conditions, but not the habitat associated with the original plant species. Restoring the conditions necessary to reestablish the natural species richness, if not the entire original biodiversity, is more likely to restore structure and processes determining material flows and cycles than native biodiversity resulting from restoration of total ecosystem biomass, production, and decomposition.

### 3.3.3 Hydrologic Cycle Outputs

#### 3.3.3.1 Water—A Material Of Exceptional Interest

Because water is a material of extraordinary functional importance in ecosystems and Corps resource management activities, it is treated here separately from other material flows and cycles. The behavior of water in that part of the hydrologic cycle influenced by living processes comprises a fundamental set of ecosystem functions (Muller and Windhorst 2000) of particular relevance for water resource management. Water is the universal solvent and the most important transport medium for nutrients, toxins, sediments and other materials. Watersheds are among the easiest ways to define ecosystem boundaries over a wide range of geographical scales and are the most practical means for monitoring and managing input-output dynamics of inland aquatic and estuarine ecosystems. Because the Corps manages surface water flow and storage, associated *hydrologic functions are among the most relevant to Corps-planned ecosystem restoration.*

### 3.3.3.2 Ecosystem Influences On Water Cycles

While hydrologic cycling mechanisms are predominantly physical and cycle fluctuations have profound impacts on life functions, life processes in the watershed also stabilize natural hydrologic fluctuation significantly. Many aquatic, wetland and riparian species are adapted to specific patterns of hydrologic variability. Through water dynamics, upland ecosystem functions influence the integrity of aquatic ecosystems. Natural communities generate surface roughness and organic soils with high capacity for retaining water and diverting surface flows into other locations, including subsurface ground waters. This in turn influences the efficiency with which essential nutrients are retained and cycled in vegetated terrestrial and wetland ecosystems, and kept out of aquatic ecosystems (e.g., Likens et al. 1977). Within aquatic ecosystems, the collective dynamics of water volume and flow are among the key physical variables contributing to the evolutionary history of inhabitant communities.

### 3.3.3.3 Corps Influence on Water

Civil works influence water and associated materials much more directly than ecosystem self-regulation and energy-flow functions. The Corps regulates surface water movement and shapes the form of surface water channels and basins. It indirectly affects the import and export of nutrients, toxic contaminants, and sediment, and determines the initial availability of these resources and inhibitory agents to ecosystem process. Without explicit linkage in restoration models, discovery of those links is left to the expertise of restoration planners.

One strength of the Corps with respect to restoration planning is its long history of hydrologic and hydraulic modeling, which can be useful in mathematically characterizing aquatic habitat. The Corps has much less experience either in modeling the effects of ecological process on watershed discharge of water into surface basins and channels (more the realm of the US Forest Service, Environmental Protection Agency, and Natural Resources Conservation Service) or in modeling the impact of aquatic habitat on community form and function (more the realm of the Fish and Wildlife Service and Environmental Protection Agency).

### 3.3.3.4 Which Is The Better Natural Integrity Measure—Hydroregime or Biodiversity?

Is the hydroregime associated with the natural ecosystem integrity a more appropriate target for restoration than biodiversity? Where hydroregime is the sole assortment of processes altered, it could be an effective indicator for ecosystem integrity once the relationship between ecosystem integrity and hydroregime is determined. But other, somewhat independent, factors often are involved such as changes in water chemistry and barriers to natural movements of keystone or dominant species. While variation in water volume, velocity, depth and width might be simulated at water control structures, uncorrected modifications can remain limiting in many situations (e.g. water temperature, nutrients, oxygen, turbidity, particulate and dissolved organic matter , bed load movement, physical barriers to organism movement).

In evaluating restoration plans, understanding of all of the significant ecological attributes interacting to determine the biodiversity is needed to assure restoration of some predetermined level of natural integrity. Restoration of all habitat and community conditions needed to reestablish biodiversity is a more reliable way to guide restoration of natural ecosystem integrity than restoration of select properties of habitats alone, even when those properties are exceptionally influential.

### 3.3.4 Sustainability, Self-regulation, and Functional Stability

#### 3.3.4.1 Functional Stability and Resilience

Odum (1971) identified functional stability as the key attribute of natural ecosystems. Restoration of *self-regulation* generally results in greater *stability* of natural function and sustained provision of natural ecosystem service. Natural functions are the source of natural services, and are expressed in a wide variety of biological and physical outputs. But the self-regulation that results in functional stability is a biological master-function that determines the *sustainability* of all physical and biological outputs from ecosystems.

Ecosystem stability is often characterized by functional and structural resilience, which is defined in two different ways (Holling 1973, 1992, 1996, Gunderson et al. 2000). Most commonly, *resilience* is recognized as the capacity to reestablish a predisturbance equilibrium condition of structure and function following moderately stressful events. This form of resilience maintains and restores functional efficiency and is measured by *resistance* to disturbance and speed of return to equilibrium. This form of resilience usually results in a structural and functional recovery sequence (called ecological succession especially by plant ecologists) that is generally predictable in natural ecosystems in which a large reservoir of native species remain in the system and serve to recolonize stressed sites, once the stress is relieved.

Less commonly, resilience is recognized as the extent an ecosystem can withstand stress before changing to a different functional and structural state—that is, to another *stability regime*. This form of resilience maintains function at another level of efficiency and is measured by the magnitude of the destabilizing stress that “flips” some fraction of the ecosystem into another stability regime dominated by a substantially different biotic community and different habitat attributes. Destabilizing stress may take the form of either natural or human caused extremes, such as intense fire, flood, storm, drought, agricultural and urban conversion, and intense and pervasive pollution. The more persistent effects often act through altered soil and sediment structure, nutrient concentrations, and toxic contamination, and through the long-lived dominant species that reestablish following the stress. Destabilization is more likely to occur in pervasively modified ecosystems, in which the pattern, age structure, and other features of the dominant and keystone species have been substantially altered, thereby changing the species recolonization composition and sequence in the stressed site.

An example of functional efficiency maintenance is a river valley floodplain exposed to regularly encountered seasonal flooding. Floodplain species either continue to function, cease functioning but persist through the flooding, or are killed or driven from the floodplain. Recovery of function and equilibration following these "routine" events usually is rapid and generally predictable as locally extirpated species return to the floodplain from nearby refuges. As flood events become more extreme and far reaching, recovery following the event is prolonged, but given time, returns to the predisturbance equilibrium state as long as the sources of recolonization in the surrounding natural ecosystem generally remain intact (not fragmented). Great enough extremes in fragmented ecosystems, however, cause long lasting changes in the environmental forces and constraints operating in the floodplain and river habitat (e.g., all of the fine sediment and soil is eroded away leaving only large rock behind) and communities. Then the riverine ecosystem locally "flips" to another functional and structural state that is, for practical management purposes, permanent. This level of function can differ substantially from original rates and efficiency of energy and material transfer and conversion. Photosynthetic efficiency and plant production, for example, may decrease significantly in a river channel scoured of all its nourishing sediment and remain that way for a very long time even in an otherwise intact river ecosystem. Yet at least some production is sustained.

These different expressions of resilience have restoration implications. Restoration of self-regulating functions would be expected to restore resilience, greater functional stability and greater reliability of associated natural ecosystem services. When disturbance has resulted in a change within the same stability regime, restoration can work readily with natural resilience to restore the original equilibrium. These relatively predictable responses are most likely to occur in areas within and adjacent to naturally intact ecosystems.

As ecosystem conditions become more generally disturbed, however, the disturbance often increases the extremity and size of stressful events (e.g., flooding and drought in disturbed watersheds) and alters the recolonizing landscape. These changes in stress and landscape increase the probability of a flip to an alternative stability regime through processes that are not very well understood or predictable. Attempts at restoration may not be able achieve the original state of self-regulatory equilibrium and the result may exhibit structure and function quite different from the planning objectives. The probability of an alternative state establishing after the stress recedes increases as ecosystems become more fragmented and otherwise modified. This second, less traditional view of resilience may be the more relevant for managers attempting to deal with ecosystem restoration issues where cultural modification is extensive and intensive (Holling 1996). As Holling (1996) asks, "*If there is more than one objective function, where does the engineer search for optimal designs*?"

The outcomes of restoration actions in highly disturbed ecosystems are less likely than lightly disturbed sites to take the form and function of the original state and more likely to result in some "flipped" version of it. A flip of this sort in the restoration process should be of no consequence if greater naturalness alone is the restoration “design objective” representing the resource of significance. Each stability regime is an equally natural result of restored ecosystem function and structure. But if the specific resources and services are intended, such as particular rare parts and processes, this flip to a new regime may fail to carry the desired service with it. In addition, policy states that the restored condition should be more like the condition that would have occurred if no human impact had occurred in the first place. Regardless of the greater naturalness of the restored process, a new stability regime resulting from human disturbance in the influential landscape, may not satisfy this goal.

The scale of disturbance with respect to the ecosystem is a critical variable determining resilience. McNaughton (1977), for example, found that communities with the greatest production stability varied most in species composition adjusting to natural climate change. In adjusting, some species drop out locally while others with similar function replace them by colonizing from outside the disturbed ecosystem area. Functional stability is maintained at the local ecosystem level while component parts are maintained at a larger ecosystem scale.

It is therefore more consistent with theory and observation to expect species richness and other indicators of biodiversity to reestablish only approximately in a disturbed fraction of the ecosystem than to expect the same compositions. Specific compositions of rare species are especially prone to unpredictable restoration. Rare species not previously present may show up in place of the previous rare inhabitants. Especially when the services of rare species are of concern, the scale of ecosystem restoration planning needs to be adjusted to account for the dynamic between local disturbances and species resources in the influential landscape. In restoration actions, the risk of restoring at least some fraction of all significant rare species in an area of degraded ecosystem increases as the total number of significant species targeted for restoration increases.

*Functional stability* influences the reliability of various ecosystem services, such as the reliable supply and safe delivery of water for navigation and consumption, the production of raw-materials for commodities and recreational use, and the provision of suitable habitat for species vulnerable to extinction. Many of these services can be reestablished locally without the same species composition becoming established. This does not imply that restoration of functional stability will necessarily increase the value of an artificially enhanced service, although it might. An example would be watershed restoration above a flood-control impoundment resulting in decreased erosion that extends impoundment service life. More likely, the summed value of restored natural services may increase enough to warrant reduction of the artificial enhancement effects. For example, the removal of a dam and levees might result in more flow variation and flood threat while the restoration is justified by improved reliability of ground water quality, status of endangered species, and outdoor recreation.

#### 3.3.4.2 The Source of Self-regulation and Stabilization: Genetic Information

Much of the important function and service associated with maintaining unique genetic information is linked with globally scarce species. In addition to potential resource-development value, those species provide functional “backup” that replaces common species when ecosystems undergo exceptional stress. Scarce species are not missed in most ecosystem functions under ordinary conditions, but are significant for sustaining natural ecosystem resilience and management options well into the future. The scarcest resources globally (e.g., species vulnerable to extinction) are among the most significant of those resources, and the most challenging to restore. Recovery of scarce species involves much greater uncertainty and risk than the restoration of common species and associated functions. This risk is often a reason given to avoid targeting scarce species, especially in small restoration projects, and instead emphasizing restoration of more common function and structure. (That rational of course misses the restoration point entirely). A fundamental way to control such risk is to scale up the recovery of ecosystem resources to a more inclusive level of influential landscape and community composition. Of course that is more expensive.

The most *critical function* for regulating and sustaining all ecosystem functions is the renewal of existing genetic information and generation of new genetic information**.** While there are abiotic self-regulating mechanisms that can act independently of biological process—such as the effect of humidity on evaporation or slope degradation on erosion rate—ecosystem resilience and self-regulation come from interactions of communities with their habitat and is imposed by inherited *genetic information*. Loss of genetic information reduces future resource development potential for various commodities, recreation, waste treatment, and other services.

Predicting exactly how much and what type of genetic information will result in significant resource change is impractical. Lacking that predictive knowledge, some benefit is derived from protecting all ecosystem processes that renew existing genetic information and generate new genetic information. Sustaining genetic information now at risk of extinction typically translates into policy associated with preventing species endangerment and recovering endangered species viability through natural ecosystem restoration. Sustaining the generation of genetic information requires maintenance of evolutionary context and process resulting in adaptive speciation at historic rates.

*Adaptive speciation* is a function that maintains all other life structure and function and *is among the least likely of functions to be artificially enhanced through engineered management.* While genetic traits of a few high-profile species can be maintained artificially in zoos or other means of last resort, it is generally accepted by conservation biologists that the variation and variability in natural ecosystem conditions that maintains adaptive speciation cannot be adequately simulated or enhanced on a comprehensive scale. At some point along a diminishing trend, the capacity for functional self-regulation and sustainability will diminish as species holding genetic information are lost at a faster rate than adaptive speciation replaces them.

The great concern existing among evolutionary ecologists and conservation biologists about decreasing global biodiversity (e.g. Wilson 1988, 1992; Heywood 1995) has loss of genetic information at its foundation. This loss has caused extensive reexamination of natural resource management actions. Native biodiversity has been eroded in numerous artificially-enhanced ecosystems (e.g. Stein et al 2000, Noss et al. 1995). Habitat degradation is the most frequently cited reason for loss of global and local biodiversity (e.g., Mac 1998). The ecosystem integrity required to sustain unique biodiversity has moved to the top of the list of management concerns (Schulze and Mooney 1993, Wilcove et al. 1998). Numerous U. S. freshwater ecosystems have undergone changes that threaten native biodiversity, but species in large rivers and isolated freshwater springs appear to be among the most threatened with total extinction.

The relative vulnerability of inhabitant species to extinction have been described for aquatic regions in the U. S. as delineated by watershed boundaries (e.g., Abell et al. 1998, Stein et al. 2000). Noss et al. (1995) have defined specific ecosystem types at risk of extinction. Such inventories typically include sensitive species not yet listed in addition to species officially listed under the Endangered Species Act. Definite "hotspots" of species vulnerability occur among aquatic regions. Scientific evidence indicates that recovering and sustaining vulnerable species involves preservation and restoration of ecosystems associated with those species. Mac et al. (1998) document the U. S. regional trends with respect to biodiversity and provide some insight into corrective actions.

Existing and future inventories of ecosystem vulnerability to species extinction may provide a basis for identifying restoration priorities. Perhaps as important, if management measures are successful in restoring the sensitive, threatened and endangered species collectively or even in part, most if not all of the physical, chemical and other biological attributes contributing to natural integrity would also be restored. Because management choices are limited by the loss of ecosystem components, reversing the trend of diminishing sensitive species and the fate of threatened and endangered species ought to be high on the list of restoration investment objectives.

As species become more rare, random events may play a greater role in determining the success of restoration. Therefore, those projects involving the greatest number of such species are more likely to succeed at some level because the risk of complete failure due to uncontrollable events in general decreases as more species are targeted in the planning objective. The relative risk is influenced by the degree to which the species are clumped within the same locations, vulnerable to the same threats, and affected by the same pathways and physical configurations of habitat.

The *best aggregate indicator* of the functional stability that sustains global biodiversity and other ecosystem services appears to be the native structural and functional biodiversity composing ecosystem integrity. If the relationships between habitat and community are indicated completely enough for habitat measures to restore the entire community, the native biodiversity indicator should also include the habitat needs of all of the globally rare species that form a subset of the local community biodiversity. No past measure of biodiversity has been totally complete across the spectrum of species and functions making up the biotic community. For practical reasons, they usually select for a taxonomic group (e.g., birds, fish) or a group occupying some fraction of the physical space (e.g., plankton, benthos).

The likelihood that an index of biodiversity will be inclusive of all significant functions and structures in ecosystems will increase with the comprehensiveness of the biodiversity measure and the links to habitat attributes. While species richness is commonly encountered in biodiversity indices, it is typically limited to a single taxonomic category such as fish and large aquatic invertebrates. Indices also are limited to the sampling framework used. For example, sampling fish or plankton in the water column may reveal little about the state of the stream or lake bottom. Indices often are time sensitive as well because species and other ecosystem features commonly change locations, form, and activity as seasonal changes occur. Because of these limitations, many indicators of biodiversity are functionally and structurally selective and likely to incompletely indicate the desired level of naturalness. *Care must be taken to assure that the measures of biodiversity, captured in relationships between and within habitat and community, are inclusive of the resources of significance*.

### 3.3.4.3 Habitat, Diversity, and Functional Stability

Decreased biodiversity often is associated with destabilization of ecosystem function and dependent natural services. Recent experiments confirm that biodiversity enhances reliability of function in a variety of ecosystem conditions (e.g., Naeem and Li 1997, Tilden 1997). The condition of the physical environment contributes to the maintenance of diversity, diversification, and functional integrity. Just as true, however, the condition of the biotic community, including its diversity, contributes to the development and maintenance of habitat for each species and for the entire community.

Because habitat and community evolve together into a functional whole, ecosystem restoration cannot be fully captured in an abiotic concept of habitat. Precisely predictive indices usually need to include elements about biological conditions in the influential environment of the restoration project. The location and connectedness of the project in the larger context of the landscape holding the source of community restoration components is pivotal in determining restoration success or failure.

Restoration models typically need to account for measures that restore the balance between *local* species *extinction* and *re-colonization* from other locations in the ecosystem. A model that links active habitat restoration measures to a passive and natural biodiversity restoration process typically will need to include landscape features that indicate habitat connection quality to colonization sources. This lack of attention to such connections is a leading cause for failure in restoring targeted species of significance in past restoration projects. Even in the best restoration efforts, attention is often too focused on restoring the hydrology and channel/basin geomorphology in a relative small segment of ecosystems. Corps policy tends to reinforce this focus. The habitat ends up being restored as an isolated island disconnected from the influential ecosystem in critical ways. With incomplete routes for re-colonization, the community fails re-colonize or to be sustained by the continuous movement of organisms and life-support materials between different areas of the ecosystem.

Restoring the needs for sustaining species vulnerable to extinction extends beyond physical habitat to the entire community-habitat complex because living organisms contribute to the habitat of other species and self-regulating mechanisms are associated with community diversity. To be effective, ecosystem restoration approached through habitat measures must carefully consider the restoration of community-habitat partnerships required to accomplish the justifying objectives. Restoration of physical habitat is inadequate for recovering genetic information held in endangered species without assurance that previous predator-prey and other community interactions will be restored as well.

Examples abound of species endangered in part because a non-native predator or competitor invades the system, or a species they depend on disappears. Invasive species, such as lampreys, combined with other factors probably played a role in the extinction of a white fish species in the Great Lakes following lock and dam construction (Smith 1972). The freshwater mussels that have undergone extensive endangerment and extinction in southeastern rivers usually require a unique fish-species host for the larval stage to survive and local elimination of that species probably has contributed to mussel losses (Williams et al. 1993).

#### 3.3.4.4.Resilience, Stability States, and Natural Ecosystem Integrity

Holling (1973, 1992, 1996) has observed that ecosystems respond to stress by shifting to a different level of integrity and functional stability once the resiliency of a particular functional state is exceeded. One variation of this concept is illustrated with a simple model in Figure 3.3. As long as environmental stress does not exceed resilience, an ecosystem will generally recover the preexisting state of equilibrium once stress is relieved. The process depends on population re-colonization from within and outside the disturbed area and natural community succession. For some ecosystems, resilience may act over decades to centuries following extreme stress.

Ecological theory contends that a naturally integrated state occurs over a range of structural and functional conditions reflecting the variation associated with ecosystem instability (Holling 1973, 1996). In any one state, some mean condition also exists short of the maximum integrity for that state. For example, as illustrated in Figure 3.4, a species richness index of 100 might be identified in a sequence of such determinations with an index range of 90 to 100 and a mean of 95. A corresponding change in functional rates may also occur, but probably not in direct proportion to changes in species richness (this is important point is discussed in detail in a later subsection).

With stress, local parts of natural ecosystems often shift from one state to another structural state while sustaining similar functionality after some species locally die out. Production and materials cycling are sustained even though certain species are locally extirpated (but not globally extirpated). The main functional and natural service difference that might be identified would be associated with the local loss of sensitive species. With extreme stress, the composition and the function can change dramatically to a new equilibrium condition, new level of resiliency, and a somewhat different measure of functional and structural integrity. This new ecosystem state might exhibit *cultural integrity* or *ecosystem health*, however, in the sense that the community maintains coherent function around an equilibrium established under new environmental constraints, if a desired mix of artificially enhanced and natural services results (Regier 1993, Costanza 1992).

Even though many of the ecosystem services associated with the natural ecosystem integrity may be diminished from this new ecosystem state, the level of cultural integrity can be sustained indefinitely with certain conditions being met. Those conditions include continued exclusion of some species and functions from the original ecosystem

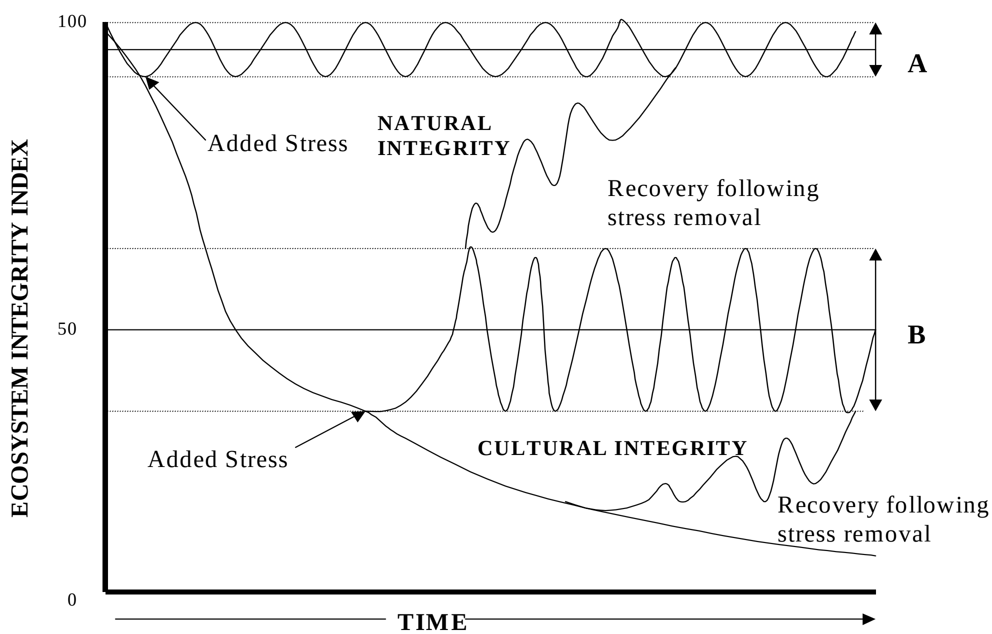


**Figure 3.4 The concepts of natural (A) and cultural (B) integrity at a local site in a river, wetland or other ecosystem. Both natural and culturally modified sites show long-term functional stability and sustainability. Stressors can be natural (usually temporary) or cultural (often sustained) but full recovery can occur from either source of stress, once removed, if enough of the natural ecosystem remains intact .**

composition maintenance of enough of the fully natural ecosystem to sustain all ecosystem parts and process beyond reach of the cultural stress. Despite local loss of integrity, this new state is self-regulating and self-sustaining, both locally and throughout the ecosystem, until environmental changes allow recovery or force a shift to another composition and level of functional stability in the stressed area. In this new state, the species richness index might average 35 and range between 20 and 50.

Evidence of change includes a different species composition, species diversity, functional rates, and variation around mean output amounts of ecological resources. Local ecosystem changes of this intensity and consistency are typical of extreme cultural influence. Examples include city harbors and riverine waterways where physical, chemical and biological change have a concentrated but local effect on otherwise natural ecosystems.

### 3.3.4.5 Resilience, Cultural Ecosystem Integrity, and Sustainability

Greater uncertainty in ecosystem condition following stress is introduced more or less in proportion to human impact in the influential ecosystem. Depending on the degree of stress, the structure and function of the ecosystem may change dramatically (flip to another stability regime), but may reach a new level of functional and compositional stability. As demonstrated in Figure 3.4, the species richness index might average only 35 and range between 20 and 50 in this new state (a species richness or other measure of ecosystem integrity could be readily converted to an index varying from 0 to 1.0 or other arbitrarily chosen range). Evidence of change includes a different species composition, total diversity, functional rates, and variation. Often in such settings, the more widely distributed species with wide tolerance to environmental conditions remain after elimination of the narrowly adapted species with more localized distributions.

Where human impacts are maintained locally the associated stress also is sustained, preventing recovery to the more natural conditions of the adjacent ecosystem. Aquatic ecosystems continuously stressed by water pollution usually reveal cultural change to a simpler structure often characterized by lower species richness and a greater functional and structural instability, such as the algal " blooms" and die-offs associated with excessive nutrient loading (cultural eutrophication). Under those circumstances the resulting oxygen depletion can cause a shift to a much simpler consumer diversity and greater reliance on microbial function accompanied by more variable community production and biomass. However, even these greatly simplified communities can persist through time, albeit with greater functional variation and less consistent delivery of natural services. They also can be restored to a condition quite similar to the natural integrity revealed in undisturbed aquatic ecosystems as long as the necessary connections are made between restored site and natural reservoirs of integrity.

Permanent structural changes that limit water-level fluctuations and eliminate biologically important connections with peripheral stream and wetland habitats often result in greater physical stability but reduced native biodiversity. Species are locally extirpated when the habitat connections and variability they require are eliminated.

However, the resulting communities also can be self-regulating and self-sustaining as long as stress, habitat fragmentation and other controlling factors including management measures remain effective. Depending on the species that are locally extirpated, the stability of at least some community functions may actually increase, such as may occur when ecologically influential migratory species are excluded from ecosystems.

If self-regulation and sustainability of function were the sole measure of *ecosystem health* many ecosystem conditions would qualify as healthy while exhibiting undesirable traits. The "open sewers" of the past exhibited community self-regulation and stability, but were esthetically displeasing and sometimes threatened human health. Even ecosystems exhibiting full integrity were often undesirable because of various perceived shortcomings resulting in corrective actions to protect human health and property and to enhance commerce and other beneficial activity.

Thus, the concept of ecosystem health is service-oriented as well as culturally sustainable. As defined by Costanza (1992), ecosystems are healthy not only when they are self-regulating but also when they produce a desirable array of natural and enhanced ecosystem services. The concept of ecosystem health marries social and ecological measures of ecosystem condition. In addition to self-regulation, a healthy ecosystem must provide a desirable mix of natural and artificially enhanced services that results in sustained level of human welfare. Neither maximum ecosystem health nor cultural integrity exists if the array of provided services is not what is desired or the system behaves chaotically.

The high probability that a number of different sustainable natural states and sustainable cultural states can exist simultaneously within a single ecosystem's geographical area indicates that sustainability of ecological structure and function is not in itself a very precise way to discriminate the relative desirability of the various states. Desired levels of service and maintenance costs are likely to differ depending on the ecological conditions and the social setting. The healthiest state is the alternative that appears among stakeholders to generate the greatest total sustained net benefit. However, social demographics and preferences may shift quickly in ways that are not easy to forecast, requiring consideration of new management measures once the expressed needs of society change. Thus, *sustaining the management-measure options* for shifting to a healthier state in response to social preference changes *requires maintenance of all of the necessary ecosystem parts and processes.*

Assurance that all of the ecosystem parts are made secure through preserving some part of the ecosystem is a fundamental priority in maintaining planning flexibility. That there is significance in this pursuit is indicated by social commitment required to maintain habitat quality critical to the viability of all rare and unique species under federal law. Ecosystem restoration is most justified, it seems, where past management decisions have compromised critical habitat and the investment risks associated with its restoration are judged acceptable. The least risk of restoration failure would most typically be associated with those ecosystems supporting the greatest number of vulnerable, globally unique species where cultural modification is not extensive and restored habitat is closely connected to existing refuges for vulnerable species.

Restoration for recovery and maintenance of vulnerable, globally unique species becomes less tenable as the degree and complexity of cultural modification increases and as the distance to natural ecosystem conditions increases. The Clean Water Act accomplished much in the way of partially restoring many of waterways, but relatively few have been fully restored and some may not be fully restorable because they have "flipped" into a state that is either technically or socially irreversible, they have already lost species to extinction, or they exist in a landscape context that is likely to replace the existing state with yet another stability regime different from the desired state. In carrying out the Clean Water Act, state and Federal agencies agreed to what amounts to levels of cultural integrity indicated by different water quality standards for different assigned uses, ranging from the most lightly used natural states to intensively used and highly modified states in urban settings. Standards for the intensively used systems may result in ecosystem conditions that "look and smell" more or less "clean" and provide some recreational fishing and bird watching, yet remain highly modified and not suitable for recovery of species vulnerable to extinction

Justification of ecosystem restoration in highly modified ecosystems might be contemplated based on the anticipated recovery of unknown levels of natural services. This justification would derive from the assumption that biodiversity indicates greater stability of ecosystem function and greater reliability of natural service delivery. However, ignorance of ecosystem relationships is just as dubious a justification for possible restoration of more reliable service provision as it would be for restoration of a floodplain for possible reduction of downstream flood damage. Biodiversity metrics alone, or any other indicator of relative naturalness and/or functional stability, do not indicate where a condition of cultural ecosystem integrity might exist or what levels of functional stability occur without prior calibration of the relationship between ecosystem biodiversity and the average amount and reliability of service provision. The biodiversities of those different states of cultural integrity cannot be predicted without prior measure of the conditions determining both the stable states and their associated biodiversities.

Understanding of the links between habitat and community in ecosystems, and to resource outputs that provide natural services, is key to restoration success. Those links determine the necessary management measures and their investment justification. The relationships and interactions need to be determined and quantified if ecosystem restoration decisions are to effectively restore a more natural state with the anticipated resources of significance. Models that define relationships between habitat and an inclusive measure of biodiversity ought to useful for environmental benefits evaluation, but only when the connection between biodiversity and societal demand for natural service are established. Under existing Corps policy, the services that most clearly appear to qualify for objective formulation for ecosystem restoration are those associated with securing resource options for the future. Biodiversity-habitat and ecosystem models may serve to guide plan formulation to attain greater naturalness once assured that the model captures all of the conditions necessary for securing the significant resource options. When specific resources are targeted, however, a biodiversity model will not indicate the significance of plan effect without prior calibration of the relationship.

### 3.3.5 Biodiversity: The Most Inclusive Output Indicator of Naturalness

If any result standouts from this discussion, it is the complexity that exists in the relationships among structures and functions comprising the interactive complex of habitat and community that define the naturalness of ecosystems once each state of naturalness is fully described along scales of human effect. As conceptualized in contemporary ecological thought, biodiversity is the most inclusive output measure of complexity in natural and humanly modified ecosystems. Measures of ecosystem naturalness in ecological output response to natural process and management measures can be indicated by holistic measures of community production and biomass, ecosystem materials retention and export, water discharge dynamics and amount, and stabilizing functions associated with resilience, and/or biodiversity. Some functional outputs are most evident in natural community process, such as biomass production and population dispersal. Other functional outputs are associated more with the physical habitat, such as watershed discharge of water and transported materials. All are interrelated sets of functions and structures that often link directly or indirectly to biodiversity as measured in studies of natural or human-caused variation in habitat and community expression.

Ecosystems are too complex to adequately characterize for restoration purposes without multi-metric habitat-community models. The most inclusive concepts of biodiversity extend beyond the community into the physical habitat (Heywood 1995). Simple biodiversity measures, such as a species richness-area relationship (Rosenzweig 1997), offer little for assessing the total ecosystem condition without more explicit links to the qualities of habitat and community conditions. The more inclusive measures are more likely to be multi-criteria indicators incorporating both habitat and community properties. Measures such as IBI may qualify, but only after the community outputs making up the index are thoroughly linked through cause-and-effect relationships to community-habitat variables.

As determined above, while biodiversity as it is measured now is the most inclusive indicator of biological naturalness, it is not a totally inclusive measure. Even if that were not the case, the biodiversity of natural integrity does not seem to hold up to the need for a national-level of “standard-unit” measure. Whereas the biodiversity existing locally in an ecosystem can be gauged against fully natural sites within an ecosystem, there is no logical way to compare across ecosystems nationally. Two ecosystems of equal integrity can have very different biodiversities. It is also difficult to determine what increments of biodiversity mean in terms of their relative naturalness, especially when the service value perceived is intangible.

### 3.3.6 Ecosystem Outputs Other Than Biodiversity Outputs

All of the above ecosystem output categories have centered on the relationships between habitat and the inhabiting community. The use of habitat-based methods has been in particular criticized for the likelihood of their not being inclusive enough of all Federal interests associated with a restored condition (NRC 1999a). Community-based habitat units as indicated by a comprehensive definition of biodiversity are more likely to be inclusive of all renewable resources that are the product of ecological process. Community-habitat measures may also be complete enough indicators of a more natural state to actually restore physical features and outputs underlying certain services (e.g., water supply, treatment, and regulation). Even so, a biodiversity indicator will not provide explicit measure of important outputs, such as water discharge, quality, and flow changes caused by life processes. Although such methods might formulate for the more natural status of these outputs, they provide no quantitative information useful for indicating relative or absolute value. In such situations multiple measures are required when they are relevant to the decision process.

For example, in addition to supporting living plants, animals and microbes through habitat functions, wetland functions influenced by life processes include groundwater flux (recharge and discharge), wave and current energy dissipation, surface and subsurface water storage, and nutrient and other materials sequestration and release. These functions are not necessarily restored independently of life functions, but the output measure that needs to be known to evaluate the resource significance is not captured in a biodiversity measure. These need to be considered in addition to community biodiversity for thorough evaluation of all effects. Ecosystem function indices such as those described by Smith (1995) are able to address the relative quantities of biodiversity and various other outputs (e.g., water discharge, nutrient retention) independent of biodiversity.

Existing measures of biodiversity now fall short of representing the nuances of natural conditions accurately and uncertainty caused by random events limits the precision of measurement. This is true for both specific biological resources of significance, such as rare species and physical outputs. While the relatively common biodiversity included in most existing models may be accurately foretold by habitat restoration, the forecast recovery of biodiversity may not include recovery of specific resources. Even if a complete understanding exists of interactions among habitat, communities, and all specific outputs from ecosystems, many of the natural processes influencing human services will remain uncertain because of random events. Thus planning needs to consider the risks of not realizing significant enough response based on averages.

### 3.3.7 Relationship of Significant Resources to Naturalness and Biodiversity

Restoration to a more natural condition can produce many different specific resources of significance and at different rates of recovery. The value of some of those resources varies with social context—for example, the value of increased discharge from a wetland area depends on the location of those who might make use of it. Therefore the unit-product value varies from one project location to another and their contribution to national benefit can vary widely. Other product values are less situational and more constant across projects nationally.

Study of ecosystem disturbance and recovery reveals that functions in general restore at a faster rate than species diversity as indicated in Figure 3.4. Most of the major functions of ecosystems restore relatively quickly following natural and moderately destructive events, such as fire, drought, flood and storm (curve A in Figure 3.5). This is the restoration process associated with the more traditional pattern of ecosystem resilience. Typically, a few pioneer species enter quickly from the adjacent intact ecosystem and restore much of the production, biomass, mineral flow and cycling, hydrologic effects, functional resilience, and functional sustainability early in the natural restoration process. Much of the structural biodiversity associated with rare species follows later (curve B in Figure 3.5), mostly adding functional redundancy to the ecosystem. In complex ecosystems, the pioneer species often become less dominant as ecosystem conditions are altered by the inhabitant community and by those species arriving later in the recovery process. Some of the late arrivals may eventually become the dominant species. While the globally rare species are most often associated with the most structurally diverse condition of ecosystems, they can also be associated with any stage along the continuum from fully disturbed to fully recovered.

This implies that those natural functions and services closely linked to biomass and production dynamics will often recover at faster rates than services associated with the rare structural attributes aligned with scarce species. Services associated with water supply for navigation, irrigation and domestic use; flood damage reduction; commercial fisheries; fish and wildlife based recreation; natural features-based sight-seeing, erosion control, water quality treatment, and carbon dioxide regulation may become reestablished for the most part long before services associated with the rarest species become established along the gradient of partial to full restoration. In at least some cases nonnative species can reestablish many of these specific functions and services associated with community production and biomass about as well or better than the native species (as discovered by the early “restoration” professionals who often “planted” non-native species to control erosion, to provide fish and wildlife-based recreation, and to “protect” watersheds.

An important natural service associated with the later arrivals to a fully recovered biodiversity is a functional redundancy service. Redundancy provides ecosystems with optional parts, which may assume a higher functional profile when conditions change in the ecosystem environment. Functional redundancy adds to the resource development options associated with globally rare species. Ecosystems provide a basic service by sustaining those options. Of course, the more globally important service that is often associated with the scarce biodiversity in ecosystems, is the maintenance of genetic information that is vulnerable to extinction. While the values of many ecosystem services depend on distributions of resource supply and demand, the value of nationally significant biodiversity does not. What usually makes ecosystems unique and rare is the global rarity of their endemic species, regardless of where they occur in the Nation. The greater the number, uniqueness, and vulnerability of species at risk, the greater is the deficiency in ecosystem value that needs to be restored. Such a metric is comparable within and across ecosystems at local and national (or international) levels. Few other measures are universally of constant value across the nation (carbon sequestration may be one), but none are, according to national policy, more valuable.

In generally modified and intensively stressed ecosystems, the relationships between restored function and structure becomes even more difficult to predict with increasing probability that a new stability regime will become established. A new regime may result in different functional and structural performance, and may include few of the rarest species associated with the desired restoration condition. The new stability regime might be suitable for other rare species, but because it is out of context with the surrounding ecosystem, those species may never re-colonize.

Restoration can produce many different specific resources of significance. The value of some of those resources varies with social context—for example, the value of increased discharge from a wetland area depends on the location of those who might make use of it. Therefore the unit-product value varies from one project location to another. However, some products are more constant across projects Nationally and are amenable national summation of benefit produced. One measure is based on the genetic uniqueness of scarce biodiversity in the form of species at risk of global extinction. What usually makes ecosystems unique and rare is the global rarity of their endemic species. The greater the number, uniqueness, and vulnerability of species at risk, the greater is the deficiency in ecosystem value that needs to be restored. Such a metric is comparable within and across ecosystems at local and national (or international) levels.

### 3.3.8 Evaluating Projects and Priority Ranking of Ecosystem Restoration

Ecosystem outputs *in themselves* provide no indication of their social significance. Except when they become very scarce with respect to social wants and needs, the total amount of community production, biomass, materials of any kind, including water, or biodiversity fail to provide consistent clues to their social significance. In aquatic systems, high community production often signals low economic value per unit of production because the more valued commodities are associated with low-production states in which the commodities themselves are relatively scarce. Wetlands, in contrast, are typically high-production ecosystems providing resources and services that are now valued highly enough *in general* to establish a national goal of no-net-loss of wetland function and area. This goal came about because wetlands were perceived to be relatively scarce ecosystems that were rapidly being converted to other uses before their *specific values* were completely determined. Similarly, the extinction of species is resisted by provisions of the Endangered Species Act, including species with no known existing use but with possible individual and aggregate value yet to be determined.

There is often no close connection between *local* biodiversity and the resource significance and value in many ecosystems. A large proportion of the world's species are

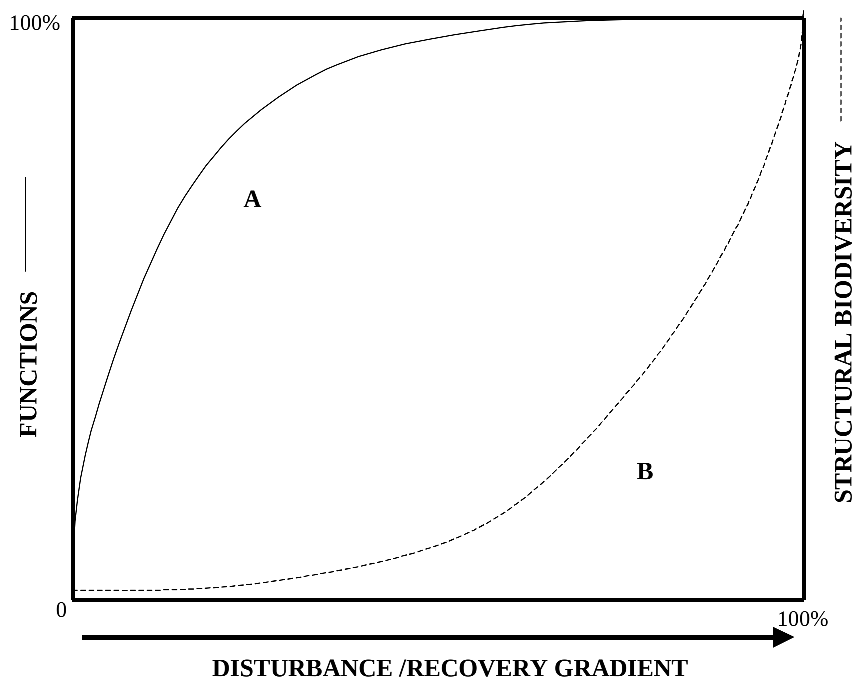


**Figure 3.5. Generalized recovery rates of ecosystem functions associated with community production and structural biodiversity following local disturbance in an otherwise intact ecosystem. Causes of disturbance include storm, flood, drought, severe fire, agriculture, dredging or other stressors. A indicates the relatively rapid rate of recovery for many functions and services associated with production and biomass recovery. B indicates the slower rate of individual species recovery, with species on the right side of the recovery curve mostly providing functional redundancy.**

not so scare they are at risk of extinction and their collective diversity has modest positive economic value, such as that associated with recreational sight-seeing and nature observation (e.g. birding). The most productive species range from low-value "weed", 'nuisance", and "pest" species to high-value "resource" species, the value being indicated less by the total supply and more by the relationship of supply to demand. Even when ecosystems have relatively high biodiversity, but composed of common species, they are not especially valued for sustaining their biodiversity independent of recreational and other such economic value, and are often converted to more highly valued uses, such as back yards and recreational reservoirs. Many substitute sites of equal biodiversity exist for recreational or educational use. In contrast, a number of ecosystems with low total biodiversity are protected from conversion to any other use than support of their biodiversity because much of that biodiversity is globally scarce.

Relative *scarcity of resources* is the key to determining most of what are held to be significant services via institutional laws, public opinion, and technical assessment suggested by the Water Resource Council (1983). Once ecosystem parts and processes become hard to get or are gone altogether, management choices become more limited. Restoration options diminish with the attrition of unique ecosystem structure and function.

Thus *a "keystone" priority* is secure maintenance of all ecosystem parts and processes vulnerable to extinction. Recovery to a more secure status ought to be high priority. The protection and recovery of endangered species authorized in the Endangered Species Act, regardless of all but the most onerous of social costs, is the primary institutional evidence of the non-monetary value attached to environmental sustainability through its living species. Held within the genetic information and traits of species vulnerable to extinction is the potential for resource development with global benefit. Loss of those traits and genetic information limits resource development and management options. Among the most important lost options is the ability to fully restore ecosystems. It is difficult to conceive of non-monetary benefits more important than the benefits associated with sustaining the rare parts and processes of ecosystems.

The potential exists for ranking relative non-monetary benefit based on the amount of genetic information and associated species traits that might be made more secure for the future. Recent studies have already been conducted primarily for the purpose of identifying ecosystems for conservation attention based on the global scarcity of their biodiversity (Stein et al. 2000, Abell et al. 1998). In ranking ecosystems with respect to the scarcity of biodiversity, the methods used by conservation biologists consider both uniqueness and vulnerability. DNA analysis and other molecular techniques can aid this process and will increase in importance in the future, but, in the near term, practicality dictates that the identification of species and community uniqueness relies mostly on more traditional taxonomic methods.

The uniqueness of a site's biodiversity is typically determined by the number of closely related forms within taxonomic categories. Final scarcity rankings are established after considering uniqueness at each taxonomic level. For example, a subspecies that is the only member of its species, genus and family is ranked much higher than a subspecies that is one among many other subspecies, in one of many species, in one of many genera in a family. Ecosystems harboring numerous globally unique families represented by one to a few species are ranked much higher in uniqueness than ecosystems made up of a few globally widespread families composed of numerous species.

Vulnerability to extinction is based on population factors such as relative abundance, distribution, reproduction rate, mortality rate and the intensity and imminence of new threats. A species classified as vulnerable implies that it is overly scarce with respect to its prospects for continued viability. A species of high total abundance that is widespread and has a reproduction rate that counterbalances mortality rate is ranked low in vulnerability compared to a species of low abundance concentrated in one geographically small location where there is little indication of successful reproduction. Biotic communities and ecosystems can be ranked based on the multiples of uniqueness and vulnerability summed to a biodiversity scarcity score. A community that has numerous species that are the sole members of their genus and family and are highly vulnerable is ranked higher than a community with few such species.

A number of government and nongovernment organizations use some kind of scarcity ranking method based on attributes similar to what is described above. Widely accepted methods used in the U. S. to rank global vulnerability of species are illustrated in the database, NatureServe (http://www.natureserve.org/explorer/index.htm), which was developed for the state natural heritage programs and The Nature Conservancy. The method is also described in Stein et al. (2000). Uniqueness ranks have undergone substantial theoretical development (NRC 1999b), but have yet to be incorporated into a national database. Management priorities for the Endangered Species Act are based in part on uniqueness as generally described above and there species are internationally ranked by the International Union for the Conservation of Nature(IUCN). Vulnerability and uniqueness classification methods require the knowledge base and skills of taxonomic specialists. They rely on a variety of classification and population evaluation techniques, subjective judgment, and peer consensus.

It is much easier to see the utility of ranking species and community scarcity in a national portfolio of restoration priorities than it is in determining whether or not biodiversity in a project area is scarce enough to consider it nationally significant. Because a gradient of biodiversity scarcity exists, there is no natural “threshold” that signals when a species or community is scarce enough to determine it is a “significant resource” as defined in Corps planning policy. The red lists of various organizations provide guidance, and, institutionally, the species listed under the Endangered Species Act would have high priority. But because many species that may deserve listing are still under consideration, a larger number of species undoubtedly qualify. Whether or not it is an artifact of status categorization is a relevant question, but the ranks of vulnerability indicated in NatureServe break out into relatively secure species in the G4 category and somewhat vulnerable species in the G3 category. In any case, determining the significance threshold will require judgment, which might be applied at the national, division or district level, depending on institutional policy.

To the extent that various biodiversity indices and process models might capture all of the habitat needs of the scarcest biodiversity, they may serve effectively to indicate the relative non-monetary benefit based on the increment of species security promised by the restoration action. Biodiversity models driven by community habitat variables are most often calibrated from the responses of the most abundant species in the system. Such models are least dependable for guiding recovery of the scarcest biodiversity. An effort has to be made to integrate their needs into the planning process, or at least to assure that their needs will be met by the needs of the more common species. This can be done by using community models and species or guild models in sequence, first formulating for the common biodiversity and then evaluating for whether the conditions provided are suitable for the significant resources. When both types of models are well calibrated, the habitat conditions indicating greater native biodiversity in a community-habitat model may be input into species-habitat models (or other model of resource significance) to determine the extent of biodiversity restoration required to restore for the resource of significance. This approach requires substantial coordination of model development and calibration through a carefully considered concept of the target ecosystem.

A uniqueness-vulnerability index falls far short of representing all value provided by natural ecosystem service. This index is based on the relative scarcity of species traits and genetic information, and it places high value on maintenance of the scarcest species for future management options, including restoration options. It places very little value on common species, despite the many ecological services that are provided by common species (They dominate the production, biomass and other ecological process underlying many services). Most utilitarian values, including NED, are associated with relatively common species, such as the species that support hunting, fishing and much other outdoor recreation. There may also be services and values as yet to be revealed that fall outside this index and the NED monetary index to value. Until those values are revealed, however, a uniqueness-vulnerability index is a good interim measure of NER contribution worthy of serious consideration.

### 3.4 Ecosystem Inputs: Management Measures and Natural Process

#### 3.4.1 Natural Process and Management Measures

The *management measures* used in ecosystem restoration projects led by the Corps and other management agencies are inputs, the costs of which are justified by the promise of a gain in desirable ecosystem outputs. The natural ecosystem inputs required to sustain ecosystem biodiversity, integrity, and associated functions and services include all of the environmental forces and constraints that operate and regulate the system from outside including the motivating energy (usually solar, gravity, and chemical energy); the topographic, geologic, and hydrologic features; and all of the associated natural biodiversity. Among these inputs some are more subject than others to cultural modification and to management measures. While some of these forces and constraints are not manageable, many are.

Because different ecosystems take their unique form and function along a gradient of environmental forces and constraints, *the functional outputs from an ecosystem often become inputs for other ecosystems.* Water flow is among the most obvious examples in the Corps management domain, as it moves under the force of gravity through subterranean routes to surface streams over channel gradients and through basins to wetlands, aquifers, lakes, rivers and estuaries. Along the way it erodes, transports and deposits an assortment of organic and inorganic materials transporting them from ecosystem to ecosystem. Most Corps management measures shape basins, channels, floodplains and beaches to create or restore interactions between water, gravity, substrate, water-transported materials, and living communities.

Project location with respect to natural influences from the surrounding landscape and random events is among the more important input considerations when forecasting the without project condition and when choosing management measures for project plans. While the Corps most commonly considers “habitat measures” as most relevant to recovering a significant natural resource through ecosystem restoration, restoration measures may also influence ecosystem structure and functions other than the organisms that inhabit the habitat. Examples include filtration of particulate materials, the absorption and biological incorporation of dissolved materials, the percolation of water to ground water aquifers, surface and subsurface storage and discharge of water, and other functions in support of ecosystem services. The incidental economic and other benefits that might be associated with these structural and functional changes need to be considered to complete the planning process.

### 3.4.2 Random Events and Ecosystem Inputs

*Random events* contribute significantly to the way ecosystems function and respond to restoration measures. Species are adapted to and sustained by the natural variation of ecosystem processes, including variation resulting from random events associated with storms, floods and fires. Ecosystem integrity depends on the maintenance of variation caused by random events, which affect the colonization success of species and the resulting species composition of the restored community. The composition of rare species is most likely to change from the original composition because of random events.

The Corps operates in ecosystems influenced by random events associated with weather, hydrology, hydraulics, and community recovery process. The uncertainty associated with random events cannot be reduced without diminishing certain desirable natural services resulting from ecosystem restoration. That same variation often is modified through installation of artificially engineered structures and functions. Partial to full ecosystem restoration requires at least some reversal of this modified state either by returning natural process or by artificially simulating the variation.

Random events introduce *unavoidable uncertainty* into prediction of ecosystem integrity at any one location and time. Accommodation of that source of uncertainty typically requires adjusting the scale of planning to a larger sphere of ecosystem connection and influence. The uncertainty associated with ignorance of functional effects, which by definition are predictable if understood in an ecosystem context of proper scale, is avoided through improved understanding of ecological process. Improved understanding of functional cause and effect usually requires searching for cause at a broader landscape scale of ecosystem investigation.

### 3.4.3 Landscape-Scale and Ecosystem Inputs

Ecosystems reveal a widening range of properties when viewed through increased spatial and temporal scales. The species in ecosystems are adapted to the decimating effects of destructive random events through their *habitat connections* to unaffected parts of the same ecosystem or other ecosystems inhabited by those species. To be effective in restoring ecosystem integrity, as indicated by biodiversity, restoration measures must address the condition of these connections at a landscape scale (Norton and Ulanowics

1992). Ecosystems are hierarchical organizations that comprise numerous interactive subsystem “units” nested within more inclusive systems defined by spatial and temporal boundaries (O’Neill et al. 1986, King 1993). In the example shown in Table 3.2, ecosystem boundaries could be defined on a scale ranging from the microbial community in an insect on a rotting log located in a floodplain wetland to the entire watershed linked to the river and wetland. The numerous organisms, logs, and other subsystems contribute to the wetland ecosystem and many different wetland, pond, stream, and other riparian

**Table 3.2. Simple example of spatial and temporal hierarchical ordering of a floodplain wetland ecosystem in a watershed context.**

| SPATIAL & TEMPORAL SCALE<br>SMALLEST → LARGEST | | | | | |
|---|---|---|---|---|---|
| **ECOSYSTEM** | **INSECT GUT** | **ROTTING LOG** | **FLOODPLAIN WETLAND** | **RIVER & FLOODPLAIN** | **WATERSHED** |
| **STRUCTURAL COMPOSITION** | **bacteria** | **insects** | **rotting logs** | **wetlands** | **rivers & floodplains** |
| | **protozoa** | **fungi** | **cypress trees** | **ponds** | **headwater streams** |
| | **water** | **algae** | **birds, fish** | **lowland forest** | **headwater wetlands** |
| | **nutrients** | **water** | **decomposers** | **sandbars** | **lakes, ponds** |
| | | **air** | **sediment** | **alluvial soils** | **upland prairie** |
| | | **nutrients** | **water** | **riparian communities** | **upland forest** |
| **TIME FRAME** | **Weeks** | **Years** | **Centuries** | **Millennia** | **Many millennia** |

subsystems contribute to the floodplain ecosystem. In turn, floodplain and upland ecosystems contribute to the structure, functions and services of a larger watershed ecosystem.

Ecosystems low in the hierarchy depend more on functional outputs from larger ecosystems high in the hierarchy. The insect could not survive without the log, which would not be there without the forested wetland, which depends on floodplain groundwater maintained by watershed runoff from a wilderness watershed. If the watershed is logged, surface runoff will increase the wetland sediment loads filling the depression as groundwater level drops and the aquatic community dies. A totally new array of species then colonizes the site. Small ecosystems come and go relatively rapidly compared to larger ecosystems depending on the events that shape them A floodplain wetland may last until the next major flood fills it with sediment, but other wetlands persist or are created in the process, sustaining the same general pattern and all of the component species. Holling et al. (1994) described in detail the relationship between event frequency and spatial scale of ecosystems for the Florida everglades.

Restoration of a filled wetland depends on the pattern of all wetlands in the floodplain, including habitat connections enabling colonization of the restored site to something like the original diversity and composition. That natural pattern can change, however, as upstream watershed and river channel conditions change from human impact, influencing both the total species pool available for colonization and the connections enabling colonization.

Ecosystem scale is a critical consideration for understanding the interdependencies of ecosystem functions and for plan formulation and evaluation of ecosystem management decisions (NRC 1986). Many of the failures encountered in past restoration attempts derive from insufficient ecosystem perspective when considering management measures. Corps ecosystem restoration activity often is based on the assumption that the establishment of the rudimentary physical attributes of habitat will be most certainly followed by self-restoration of the community. "If we build it, they will come", is too often the naive philosophy behind a provincial approach to restoration. Habitat restoration measures often amount to reshaping a basin or channel to more natural lines, supplying it with more-natural water-flow variation, perhaps seeding or planting it with one or two native plant species, then leaving it for nature to finish the job. The assumption that restoring a more natural physical state will assure the recovery of the justifying resources of significance because of the many pathways and uncertain processes by which unsatisfactory results can occur.

Restoration projects usually fail when the connections of the proposed habitat and community to other natural ecosystems are not restored in proper regard for all of the recolonizing organisms. Common setbacks for small restoration projects include destruction of restoration measures by "pest" species, disease, floods, droughts, wind, fires and other natural events. A significant service of integrated ecosystems is natural pest control. A freshly planted field without a full complement of predators and competitors is a banquet in waiting for the first hungry guests to arrive. The landscape scale and context considered in the restoration process can open or close ecological doors determining success and failure. A riparian restoration planting surrounded by natural vegetation harboring diverse predators is more likely to succeed than one surrounded by unvegetated terrain. A diversity of plantings of different size, species and distributions also can encourage more diverse colonization early in the natural restoration process.

At small restoration scales, onus is placed on the ecosystem manager to identify the many connections that need to be made to the larger ecosystem context to assure a specified level of restoration. At larger ecosystem scales, many of those connections become part of the internal structure and process, and less likely to be overlooked. Random climatic and biological events can overwhelm ecosystem restoration measures more often at small restoration scales. The effect shows up when small restoration projects are "wiped out" by a single storm event or even a busy beaver. In a larger restoration action only part of the project would be affected by the same random event and more comprehensive planning would provide for local recovery from an adjacent preserved or restored area.

Existing ecosystem models are most often least effective in identifying the proper ecosystem context and habitat connections that will serve as the source of colonizing species. This is especially true of simple index models, although some are conceptually better than others. They also are incapable of assessing the uncertainty of recovery as proposed in objectives. These missing elements are left to the professional judgment of planners who, because of many pressures, tend to narrow focus and assume “if you build it, they will come”.

*The concepts of habitat, biotic community, and biodiversity are scale dependent.* The biodiversity of an acre of restored habitat depends on how that acre of habitat is situated within the larger area of that same habitat, and with respect to all other separate habitats and their associated communities in the inclusive landscape. Numerous larger species derive their sustenance from a number of different habitats with different biotic communities. When present in the community, they may have a dominant effect, such as many migratory mammals, birds and fish. For a far-ranging species, the community biodiversity indicative of supporting ecosystem integrity and the effect of the far-ranging species often extends over several discrete communities and habitats. Some of these species are keystone, having disproportionate community effects. Measures taken in one habitat may not have the intended effect if the support integrity of any other subsystem also is impaired. Ecoregional determinations of vulnerable species (e.g., Abell et al. 1998 and Stein et al. 2000) include some species that range well beyond ecoregional boundaries, such as migratory fish and birds. Whether or not local measures will be effective in restoring integrity, including species at risk, often depends on the status of ecological limitations in all habitats and communities influencing species viability.

### 3.5 Section 3 Summary and Conclusions

Environmental benefits analysis for Corps ecosystem restoration projects seeks identification of widely applicable non-monetary indicators of environmental value consistent with Corps policy. It requires definition of relationships between habitat and inhabitants to forecast indicator response to restoration plans, including a no-action plan. Ordinarily, Corps policy limits the choice of indicators to measures of function, structure, and dynamic process that reflect the condition of socially significant resources and are consistent with a more natural ecosystem condition. Ecosystem restoration benefits are to be measured in terms of changed resource quality that is a function of habitat improvement. Because habitat is defined by the living species and communities that inhabit them, the resources to be measured are expected to be the product of the living inhabitants—species, communities and their effects on the physical environment.

Because some benefits from restoration may derive from the purely physical effects of altering topography and hydroregime, it is likely that evaluation based on inhabitant effect alone would not capture all value in the Federal interest, as noted by the NRC (1999b). However, when done completely according to Corps policy the sum of all significant monetary and non-monetary effects resulting from the project should be considered, regardless of whether they were objectives of plan formulation. This should capture all significant effect in the Federal interest based in the material changes that occur in ecosystems, but it is unlikely that any single widely applicable non-monetary measure would cover all of those effects. Any socially significant effects that are not linked to the material consequence of restoration would not be included however. Thus, for example, the value perceived by some in the removal of an engineered structure alone, without concern for what material results in the ecosystem, is independent of ecological concept and measure.

In some cases greater naturalness of community-habitat complexes may provide in itself the service of significance, but in many other cases only a subset of resources may be recognized as significant. Corps policy requires that all significant benefits and costs accruing to restoration plans be considered in the local and national perspective for the selection and recommendation of a plan. Thus, in addition to the biotic resources of significance, the value of all other biotic and abiotic output from restoration should be considered. Restoration can be total or partial, but the desired result is a self-regulating, sustainable output of resources that provide significant natural service.

Ecological concepts pertaining to measures of naturalness and to individual living resources and their products in ecosystems include ecosystem structure and function, biodiversity, ecosystem integrity, production and other energy flow, material flow and cycling, self-regulation, sustainability, resilience, and redundancy. These are related ecological concepts, but vary enough from one another in response to restoration that they cannot be comfortably considered either as one or independent in project formulation and evaluation.

Ecosystem integrity is a promising concept for guiding the restoration of ecosystems to conditions with less net human effect, or greater naturalness. In practice, ecosystem integrity is indicated by the biodiversity and physical features and processes that occur in along a gradient of human effect in reference ecosystems. Broadly defined measures of biodiversity are the most inclusive measures of ecosystem integrity, and also appear to be the most inclusive measures of the resources targeted for restoration in Corps policy. However, ecosystem integrity has little meaning outside of the context of ecosystem reference conditions. A unit of natural integrity, measured by some increment of biodiversity, has no universal meaning across ecosystems, and, as now conceived, cannot be summed in some meaningful evaluation measure of contribution to ecosystem integrity accross ecosystems at a national level. Thus it seems to fall short of a measure adequate to NER evaluation needs.

The long-term continuity of function and structure in an ecosystem, or ecosystem sustainability, is often linked to naturalness and an intent of ecosystem restoration. Sustainability can occur in a wide variety of ecosystem configurations, however, including various levels of natural configuration and human effect, which, when desired by humankind, are known as cultural integrity. Sustainable conditions can result in very undesirable states, as well, such as the repulsive and unsanitary conditions of a river heavily polluted with human wastes. Thus sustainability in itself has little meaning as a measure for NER contribution. Like ecosystem integrity—whether natural or cultural—to which it is closely related, the sustainability added to NER by each project is only meaningful in a context defined by the desirability of the outputs provided. Because both preferences and structural figurations not only can but probably will change through time, ecosystem resources and services will vary in desirability through time. Establishing a constant structure and function at a project site, even if it were possible to sustain, would not be valued nearly as highly as future management flexibility that is responsive to preference changes while it maintains all future management options.

The restoration of and sustainability of future options requires a comprehensive landscape perspective that reaches to the entire ecosystem. The biodiversity of landscapes represented in the patterns of natural ecosystem reserves and their connections to restoration project areas is of critical importance in determining the success of restoration plans. Choosing which resources are most important to restore for maintenance of management flexibility, including future restorations when desired, is most determined by the distribution, vulnerability and uniqueness of scarce resources in the landscape. These are among the most significant of ecological resources. If not considered at a landscape level, overlooked effects and random events will ensure that substantial irresolvable uncertainty will remain in the restoration process, especially when the resources are very rare and the project area is very small.

Thus restoration of the most significant resources, based on relative scarcity, becomes particularly risky at sites embedded in highly disturbed ecosystems and landscapes. Managing the risk requires information about the relationships between the ecosystem needs of the significant resources and the degree of naturalness planned by ecosystem measures In contrast, the restoration of the common resources is relatively easily assured. Ecosystem functions and associated services such as production, biomass accrual, sediment control, nutrient sequestration, and green space development, and their sustainability are relatively easily recovered with recovery of the common contributors to biodiversity. These functions and services may be in short supply for local desires but are much less likely to be so scarce as to satisfy a national need.

Common biodiversity measures indicative of ecosystem naturalness and integrity are more useful for formulating for the most common natural conditions than they are for evaluating restoration effects on scarce resources at either a local or national level. But only when they are combined with condition measures for the significant resource. Partial restoration is especially unlikely to forecast response of rare structure if the common structure and function is restored at different rates, as seems to be indicated for many of the ecosystem conditions so far studied. In many cases, the rare species structure of ecosystems recovers much later in restoration process than many of the ecosystem-level functions (e.g., production, resilience, material flow and cycling) indicated by more common contributors to biodiversity. Because very few ecosystems have are unaltered to at least some extent by humans, very few can be entirely restored. However, if the ecological requirements of the scarce resources are known, forecasts of the ecosystem conditions sustaining a more natural biodiversity, can be used to evaluate the suitability of expected conditions for the resources of significance.

There appears to be no single non-monetary unit that is widely applicable for environmental benefits measure. No ecological output reveals more possibility as a universally applicable basis for non-monetary measure of service benefit than the energy in ecosystems. Energy is the most universally distributed natural resource found in ecosystem form and process that can be compared as Joules, calories, dry organic weight energy measure. Like naturalness, they provide little insight beyond power supply into natural or management-enhanced ecosystem service and values. A related concept of power maximization has been proposed, but remains obscure and peripheral.

A somewhat less universal resource, but found in all life process of ecosystems, is genetic information. It is the "blueprint" information needed to renew life through reproduction of the variety of form and function defined in biodiversity. Virtually all services rendered by life processes are defined and sustained by the genetic information held in an ecosystem context. The amount of genetic information is most usually indicated by measures of biodiversity—most often species richness. While efforts are made to account for all species, no community-habitat indexes are nearly so inclusive. Like energy and naturalness, genetic information in itself provides little insight into many of the natural and management enhanced services and values that depend on it. Like calories, the service rendered by genes depends on its expression in form and function, and that expression has to be calibrated against social recognition of its significance to define service and value.

Assuming that scarcity is an important criterion, one of the clearest categories of specific ecosystem output indicating resources of *environmental* significance are the threatened, unique traits held in rare species at risk of extinction. Until those traits are defined clearly in terms of their full service capability and value, their restoration and maintenance sustains resource-development possibilities, including ecosystem restoration options, which would be lost with extinction. Until science informs better, each gene of unknown potential holds equal option value, and a genetic or species-based currency can be conceptually based on uniqueness and scarcity of genetic traits (NRC 1999b). This currency would have little meaning otherwise. It would misrepresent the many resource values based on active utility of ecosystem resources. The common traits found in many plants and animals economically valued for their commodity and recreational use would have low increments of environmental value as indicated by this measure. While preliminary assessments of vulnerable species and their home ecosystems provide a good start, the development of a "currency" based on the scarcity of unique species traits and genetic information is incomplete and requires further investment.

Many other *environmental* service values (benefits) are affected by the restoration of ecological resources. Certain cultural resources may fall into this category, but are not the objective of ecosystem restoration, as defined in Corps policy, which precludes cultural and aesthetic attributes of the environment. The fish, wildlife, plant and other natural features underlying recreation may serve as nonmonitary indicators of value, but seem to be considered economic values rather than environmental values in Corps policy. Other non-monetary measures are possible for other services, such as those associated with water supply and flood damage reduction. But they too are typically considered among economic values. Regardless, however, once a restoration project is evaluated for its biological resource effect, Corps policy requires all monetary and non-monetary costs and benefits to be considered in evaluation. While the decision to restore may be based fundamentally on non-NED benefits becoming reestablished in significant amounts, all national benefit and cost effect is to be considered in the analysis. The findings here are consistent with the NRC (1999a) judgment that habitat-based measures of restoration benefit used alone are likely to under-represent the Federal interest.

# Section 4.  Models For Ecosystem Restoration Planning

## 4.1 Corps Planning Needs and Useful Model Attributes

Many types of quantitative models have been developed to indicate ecological response (outputs) to natural and managed changes in ecosystem conditions (inputs). They vary widely in structure, assumptions, and ecosystem restoration planning utility. To be most useful for Corps planning purposes, ecosystem restoration models need to facilitate planning process that is consistent with Corps planning and ecosystem restoration policy. The most basic need is a model, or models, and methodological structure that organize ecosystem information so that it can be used to evaluate the effect of natural events and management measures (model input information) on ecosystem outputs indicative of both naturalness and associated significant resources.

Based on policies summarized in Sections 2 and 3 of this report, the most useful ecological models would be able to characterize the 1) existing degraded ecosystem condition, 2) the full range of more natural structure and function associated with partial to full restoration, 3) the *significant* ecosystem structure and function associated with partial to full restoration of naturalness, 4) the net changes in significant structure and function "in the planning area and the rest of the Nation" (from ER 1105-2-100), and 5) the sustainability of result over the long-term.

By Corps policy definition, the outputs representing environmental quality need to be ecological, meaning they should be the product of life processes, in total or in part. Therefore model development and choice should consider the influences of both the community and the habitat attributes of ecosystems, which interact to determine ecosystem output in its diverse expression.  But, in addition, the more useful models will also consider controlling influences that arise proximally and remotely in the surrounding landscape, often well beyond the habitat-community complex in the project area.  The influence of watershed conditions on lakes, rivers, wetlands, and coastal zones is the most usual generic example.

While the emphasis here is on *theoretical mathematical models*, ecological models do not necessarily have to be theoretical, mathematical or computer operated.  Quantitative models may not be required for evaluation where and when existing *natural reference* conditions clearly provide a *physical model* that "maps" the desired outputs through restorative measures in closely connected but degraded areas.  For restoration purposes, physical models are rarely small scale "mock-ups" of the real thing.   Most often the physical models are photographs, maps, and other representations of the desired *natural reference condition*.  These can, in very specific conditions, clearly enough inform planners about the relationships between input measures and resulting ecosystem condition that there is no further need for mathematical models.  Such clarity is typically rare, however, and good theoretical mathematical models add communication rigor, analytic flexibility, and model portability to the planning process in ways that typically elude physical models.

In addition to theoretical mathematical models, *statistical models*, which are *empirical* quantitative models, can be very useful in some circumstances, especially in situations where precision of prediction and uncertainty and risk analysis is very important and a sufficient history of relevant data is available. They develop measures of relationship between and among variables based on assumptions about theoretical models of variation and sampled-data distributions. They may be particularly useful in close conjunction with natural reference conditions, when it is possible to extend the specific conditions found in the natural reference condition into closely connected adjacent areas that have been altered.

Characterizing the more natural condition is only one aspect of modeling need. More natural structure and function of *recognized* social importance — the *significant* ecosystem resources— must be associated with naturalness to justify the investment. To be complete, ecosystem restoration planning models must identify at least two measures of ecosystem quality. One relates to satisfying the ecosystem restoration purpose, which is to restore ecosystem *naturalness.* The other relates to satisfying the need for a sound Federal investment, which is to restore ecological resources of recognized *significance*. These qualities may correlate closely in response to natural and managed influences on ecosystem performance, but often may not, as suggested in Figure 3.5 of Section 3. The *functions* supporting many natural services are likely to restore more quickly than the *structure*, which often includes the scarcest resources of greatest significance in an ecosystems biodiversity.

For greatest utility, ecosystem restoration planning model outputs need to capture both ecological resource quality and resource quantity. Corps policy indicates that the models need to characterize ecosystem quality and quantity through either a direct measure (physical units) or an indirect measure (indexes). Most restoration methods and some models are geographically based using maps of features that broadly determine habitat features and outputs. For Corps restoration projects, habitat dimensions are typically determined by water level in a channel or basin context of specified topography. Habitat area is determined, for example, by the boundaries of average water level in a river, by adjacent floodplain area in the channel, or by some fraction of wetland area within the floodplain. In coarse-grained models, the maps typically represent annual average, highs or lows, or other dimensions most relevant to the significant function and structure of inhabitant communities. Potentially useful methods recently developed track changes in habitat area through time based on the dynamics of hydrologic inputs, such as river discharge, in a topographic context.

Of course, geographic area and quality of habitat are related. The boundaries of habitat are determined where habitat qualities become so poor the space is uninhabitable. The dimensions and arrangements of different habitat attributes contributing to the environmental quality often vary with the geographic area included in a project. Boundary definition is clearest where the transition from habitable to inhabitable is sharpest, as it is at the water's edge. Within habitable space, habitat is rarely of constant quality, either within or between habitat patches. Characterizations of relative quality have been much more difficult to address. Most habitat models focus on output indicators of habitat quality, the outputs of which are then coupled with acreage (or other geographical measure) determined from maps of plan-affected area based on some prescribed method/protocol. One of the most widely used of these methods is the Habitat Evaluation Procedure (HEP).

Model portability and "generalizability" are valuable attributes for Corps planning process. All models loose prediction accuracy as they are moved from one site to the next, however, if the new sites were not among those used to calibrate the original model. While a "one-size-fits-all" model is very process efficient if justifiable, the diversity of ecosystem and planning conditions thwarts such aspirations. Empirical models (physical and statistical) are especially limited in this regard because they are typically unique and applicable to the specific site of development. Theoretical mathematical models are typically more portable as a group, but also vary among model types.

Also unlike empirical models, theoretical mathematical models can be useful ways to organize new information incrementally based on lessons learned in each planning and implementation process and on experimental research. The better models, in this regard, act to integrate empirically established fragments of understanding by bridging remaining information gaps with field-testable possibilities. The most progressive management programmatically integrates empirical and theoretical approaches through a process of adaptive management (Walters 1986, Walters and Holling 1990). In this way uncertainty due to ignorance is gradually reduced.

Inherent uncertainty in forecasts will always remain, however, because of the importance of apparently random process in ecosystems. However, no commonly used management models have dealt with this issue much, let alone well. To some extent, uncertainty can be managed by increasing model scale and by choosing more integrative indicators of ecosystem output. Ecological effects of random events often exhibit consistent patterns even though specific distributions of effects vary widely and unpredictably. For example, the fraction and general pattern of wetland and upland areas in floodplains tends to be consistent even though the spatial distribution of wetlands and uplands may change remarkably following flood events. A small scale model that implies long-term sustainability of a specific wetland because it ignores the formative context of flood events flies in the face of geophysical and ecological reality. A large scale model that indicates the general pattern and fraction of wetlands and uplands in the entire floodplain controls for the uncertainty associated with specific distributions and is more likely to indicate the more important aspects of *resource sustainability*. When the models selected for use are small scale and the controlling dynamics are large scale, much more of planning responsibility rests on the methods used to properly interpret model outputs in the landscape context.

Also related to model scale, the most useful planning models would reveal the net changes in ecological resource output quality resulting at the National level as well as at the local planning level. This requirement for a National perspective in evaluating management effects broadens the spatial scale of planning perspective needed to determine the NER contribution to the nation. This broader perspective accounts for the degree to which the significant resources may simply be redistributing in the landscape without increasing total national output (analogous to the relationships of RED and NED). It addresses the possibility of ecological influences operating outside the planning perspective that could result in resource shifts within an ecosystem without any net increase in national ecosystem restoration benefit. In worst-case circumstances, resources could be shifted to a more risky habitat situation, resulting in a net loss of significant resource. For example, rare waterfowl identified as resources of significance might simply move from one migratory habitat to another, without significant gain in waterfowl numbers, but be exposed to greater hunting mortality. Such effects can operate at a small scale as well, especially in landscapes undergoing rapid changes, such as urban development.

Models with greater spatial inclusiveness also are more likely to reveal the ratios of local and national benefit to the investment costs. For example, if the desired resources are expected to double locally, but increase by a very small fraction of 1% nationally, the information provides insight into the relative local and national scarcity of the resource, which may be a consideration in justifying the Federal investment. The value of this small incremental gain is greatly dependent on *resource sustainability*, in this case indicated by local population *persistence*, which in turn depends on dynamics in the influential landscape. Examples might include restoring vernal pools for amphibians or fairy shrimp in privately owned watersheds that are rapidly becoming urbanized.

### 4.2 Attributes of Index Models and Actual Output Estimation Models

Quantitative models fall into two basically different output categories: *relative output estimation models* and *actual output estimation models*. Relative output models express model output as an "index" of the ecosystem output of interest —typically a habitat suitability index (HSI) for Corps projects. Actual output estimation models express model output in physical units that are intended to match the actual ecosystem output measured in the field. Examples of such output include water discharge per acre of restored wetland, numbers of juvenile birds raised to migratory staging per acre per year, or average plant biomass produced per acre per year. Planning policy allows either category of model to be used.

#### 4.2.1 Relative output estimation models

*Relative output estimation models* typically take the form of species-habitat, community-habitat, biotic integrity, and functional capacity indexes. They define indexes of relative quality that are anchored in some optimal condition of maximum quality and varies downward toward zero as conditions change from optimum. Most "index" models useful for ecological assessment determine relative quality by some measure of species or biotic community output performance in a variety of habitat conditions varying from optimum to intolerable. For some indexes, the optimum condition is defined to be the most natural condition. For other indexes, the optimum condition does not necessarily have to be a more natural condition. The optimum habitat condition is defined by the maximum species or community output performance—usually some measure of abundance--which is assigned a maximum quality index value. The usual range of index values is 0 to 1, but any range can be specified.

Examples of biotic output measures include changes in population density of a species, population recruitment rate, species richness, functional capacity, and biotic integrity. Examples of relative measures of population density include bird calls heard per half hour and fish caught per 100 meters electro-fished. Species richness is estimated based on the number of species observed per unit of standard effort. Functional capacity is mathematically specified in a variety of ways, depending on function. One example, is the relative water storage capacity of an ecosystem compared to its most natural state. Biotic integrity is based on a suite of community performance indicators varying along a gradient from least human impact to most human impact. Conversion of measures to an index allows two or more different measures, including action estimates, to contribute to the calibration of an index, thereby making use of more information. Indexed qualities typically cost less to estimate than actual estimates. Being indexes, however, relative measures of biotic performance often incorporate unreported variation from sources other than the performance measure of interest.

Index models of species and community performance quality typically are structured independent of ecosystem area and need to be adjusted to make more meaningful comparisons among areas of different geographical size. This is done by normalizing geographical area to some standard unit of measure typically smaller than the area to managed, but large enough to incorporate most size related effects into the index of biotic performance. A commonly used unit is 1 acre. Quality indexes and geographical area are "integrated" by multiplying unit area (e.g., 1 acre) by the unit quality index and summing the multiples. One example of the product of this multiplication is the *habitat-unit* of HEP (FWS 1981), which in ideal circumstances can be compared directly to other habitat units of different spatial quantities and quality index values. This method relies on the assumption that a correction can be made through best professional judgment if there are important interactions remaining between the size and arrangements of geographical units and the quality of biotic performance. Where such interactions are common and intense, the utility of index models diminishes as more reliance is placed on professional judgment.

While they are usually less expensive to develop and apply than actual output estimators, relative output index models can incur unforeseen planning costs later in the planning process. As more nonlinear relationships and sharp inflections are incorporated in output indexes, the cumulative summation of "eco-units" becomes a less reliable index to total ecological output and complicates cost effectiveness and incremental cost analysis.
To be meaningful in tradeoff analysis, stakeholders need to be familiar with at least one condition along the gradient of relative quality so they can relate it to the projected change in index value. Stakeholders have an increasingly difficult time relating the change in indexed amount back to some reference condition that is meaningful to them. When these kinds of quality and quantity interactions are believed to be important, some form of ad hoc "adjustment" or "weighting" is required of the stakeholders, making the

model index meaning that much more difficult to interpret or to reproduce in similar conditions elsewhere.

A common field assumption that relative-output index models are more portable than other mathematical models can lead to erroneous conclusions about output amounts in plan evaluation and tradeoff analysis. It is often possible for an optimum index condition in one ecosystem site to produce much lower or much higher actual outputs in other situations. The index is most reliable for the conditions for which it was calibrated. Frequently, however, the calibration conditions for the original model form are not clear. As for any model, the need for calibration grows as conditions vary from the conditions for which the model was developed. Model calibration and verification ought to be based on the same performance indicators (e.g., bird calls per minute, fish caught per 1,000 $m^2$) used to construct the modeled relationships between input variables and output index. As much as possible this requires that the performance measure is taken under the same conditions for which the model was developed. Otherwise contextual variation (e.g., different seasonal and habitat conditions) can have important effects with misleading results.

### 4.2.2 Actual output estimation models

*Actual output estimation models* typically take the form of physical models, statistical models, and process simulation models. They generate model outputs that indicate actual ecosystem output amount or rate expressed in physical terms (e.g., discharge, biomass production, number of nests). Actual output estimations, make evaluation of model forecasts simple because real-world outputs can be compared directly to model output.

Physical models are small to full scale representations of the ecosystem state. While artificial models might be used to assess simple physical effects, such as vegetation effects on soil erosion (using artificial vegetation), most physical models are natural reference conditions of some kind. Small scale physical models are commonly used to evaluate ecosystem-level concepts, such as the response of vegetation plots to control of grazing, or the response of simulated rainfall runoff to vegetation cover. Such “pilot study” experimentation can be useful for testing ecosystem restoration techniques, such as plot response to restored elevations, substrate material and/or hydrology.

Full scale natural reference conditions often make excellent models for restoration, without any need for mathematical models. For example, a proposed restoration involves restoring downstream conditions to conditions like those upstream by 1) restoring the channel to a configuration like that upstream and, where possible, within the remaining outlines of natural channel in the project area (both sources of information are physical models), 2) by restoring diverted flow back to the channel (relying on the upstream condition of flow to indicate proper flow downstream, and 3) restoring a fish species of special status to the downstream habitat through natural colonization once upstream diversion impediment is eliminated (the presence of the fish species is a key part of the physical model). While transferring the model conditions to the project area may involve photographs, maps, and written specifications, no mathematical model is used in the process. While physical models have limited use under the described conditions, they are not discussed much further here.

Statistical models derive their structure inductively from samples of variables observed at specific locations. They summarize the behavior of variables in samples from the total universe of sample possibilities and estimate the range of variable behavior that might be expected among all possible samples taken. Statistical models do not identify cause and effect relationships. They simply describe the degree of relationship existing between or among variables. They often are used in combination with small-scale or full-scale physical models to characterize a mean value and variation in forecast output. Statistical models provide a measure of variation around the mean value, which can be expressed as a probability band within which the true mean lies.

Statistical models are used to test hypotheses and to extrapolate findings to a different time or location (forecasting). Hypothesis testing is used to determine whether one site condition differs from another site condition either in time or in space. Samples from a project site might be compared to samples from a reference site (the physical model of desired condition perhaps) to determine if the sites differ with respect to sampled parameters.

Statistical models are strongest for *prediction,* but only as long as the conditions they are calibrated for are clearly understood as cause and effect relationships and the context for restoration is very similar to the reference conditions characterized. As the ecological context changes, the prediction precision of statistical models tends to decrease rapidly to levels seen in other types of models, and they loose their prediction advantage. Statistical models are typically among the least portable but among the most useful when the precision of forecast result is desirable to know and to control. Because precision is a function of sampling intensity, their cost is a function of the precision desired. Process models also can include measures of confidence (or uncertainty) in the output estimate. A cruder sense of uncertainty can be determined for physical models as more natural references are visited.

Statistic models have provided much insight into the development of theoretical models and related research, but few have been used in ecosystem-level analysis. They have, however, been used to great advantage by the Corps and others for predicting river discharge based on long histories of discharge measurement at USGS monitored stations. Such databases rarely exist at the species and community levels of resource output from ecosystems. A large library of suitable references is available and they are not discussed much further here.

When there is no ecological interaction among habitat units as they are added, outputs estimates can be based on an “average” acre (other unit of geographical measure) of habitat or ecosystem output multiplied by the number of expected acres, such as 2 black ducks per acre of restored habitat for a total of 25 black ducks over 10.5 acres restored. Whether or not the areal dimensions and quality are the same for each unit, cumulative summation is relatively easy as long as each of the units are functionally independent, such as they might be for relatively small species in relatively large areal units of ecosystem. However, the ecological output per unit of ecosystem added often varies in practice as the quality of each added unit varies and sometimes it makes more sense to develop units of variable size.

Even if the ecosystem units are not independent, some spatially explicit process models are capable of capturing the quality changes that occur as units are integrated. Estimates of actual "physical-unit" output facilitate easy evaluation of cost effectiveness and incremental costs for different plans, and make tradeoff comparisons much clearer (e.g., 2.5 ducks/year verses $100 per year in water storage benefits). The primary disadvantage of these models is the difficulty often encountered in linking the specific outputs of interest back to fundamental indicators of production, biomass and numbers. Development and calibration costs often are relatively high. The main disadvantages of actual output estimation models are the primary advantages of the relative output index models.

Process simulation models provide many advantages. They have no inherently better predictive attributes than other models, and less so than statistical models, however. Because they are more explicit about process their workings are more transparent (to those who know the model language) than other models and they often make superior communication models among technical specialists. Unlike other types of models, they produce multiple outputs simultaneously and incorporate time-dependent feedback interactions that are hard to capture in index models and statistical models. This lends exceptional comprehensiveness and flexibility to their use. It is possible to link individual modules simulating the dynamics of resources of significance to a module designed to simulate a range of conditions along a gradient of relative naturalness. In this way the response of any number of resources of significance be generated simultaneous to the generation of measures of native biodiversity or other measures of naturalness (e.g., sustainability, resilience). Uncertainty due to random events can be built into the more sophisticated of such models (stochastic models). Some prototype process models are spatially explicit, providing outputs in mapped form.

Several weaknesses of index models are better addressed in models that estimate actual output amounts. Process models are especially useful in situations where many outputs are simultaneously of interest and time-dependent spatial interactions are important. This is usually the case in restoration proposals where many ecosystem alterations have occurred or are likely to occur and where a "shared vision" procedure tradeoff analysis is desired. Because they are superior models for organizing information into clear cause and effect pathways, and are particularly useful for sensitivity analysis, they are especially useful for adaptive management purposes. Process models show the greatest potential for generating integrated outputs of all NED and NER measures considered in multipurpose studies.

However, while many process models have been developed for research purposes, relatively few have been developed for management purposes. They usually require more time to assemble and/or to calibrate than index models, and tend to be more mathematically complex than other models. Their cost is typically higher than simple species habitat suitability models, but more comparable to community and ecosystem index models and to complex statistical models.

## 4.3 Important Models

### 4.3.1 Species-based Habitat Indices

Models with the longest history of Corps use are the single-species habitat suitability indices (HSI models), which were originally developed for mitigation analysis before there was a Corps ecosystem restoration purpose and NER objective. Unlike ecosystem restoration policy, compensatory mitigation policy does not require restoration of more natural conditions and habitats can be created to provide optimum conditions for species Single-species habitat suitability index values are maximum when an optimum condition exists for the species. The optimum condition for a species and the naturalness of the host ecosystem targeted for restoration may not coincide. Without knowledge of the relationship between the index value and the relative naturalness of the ecosystem, there is no way to confidently use such models to guide *restoration* to a more natural condition. Habitats can be *created* to desired levels of habitat optimality, however. Especially in situations where restoration is "simulated" through engineered means and natural conditions are not certain, single-species models are prone to guide development of a *created*, more *optimal* condition that is substantially different from a condition of greater naturalness.

HSI models are closely associated with development of the Habitat Evaluation Procedure (HEP) and, to a lesser extent, the Instream Flow Incremental Method (IFIM) developed under the lead of the U. S. Fish and Wildlife Service (FWS 1980, 1981; Bovee 1981). HEP typically was used for species in habitat settings other than flowing waters. IFIM was developed (Bovee 1981, Orth 1987, Nestler 1993) for aquatic species inhabiting flowing waters usually situated below water control structures where discharge is managed. More recently (Rubec et al. 1998 & 1999, Coyne and Christensen 1997), the National Marine Fishery Service has adapted habitat suitability measures to oceanic habitats.

HSI models were rapidly developed in the 1980s, in response to the need for evaluating compensatory mitigation determined under the National Environmental Policy Act (NEPA) of 1969. HSIs have been developed for many vertebrate species and some invertebrates. About 150 single-species HSI models are posted on US Geological Survey web pages and over 500 are believed to have been developed at one time or another. They vary greatly in quality, documentation, and the extent they have been verified and validated. Fewer models were developed for important ecological support species (mostly forage species) or for species indicative of certain ecosystem conditions.

The target of compensatory mitigation is very different from that of an ecosystem restoration target representing scarce resources in an unsustainable (degraded) state of degraded natural integrity. Firstly, compensatory mitigation did not require that a more natural condition be restored. More flexibility was allowed by accepting in mitigation the creation of new habitat optimum for selected species. The first species HSIs targeted relatively abundant species of high recreation and commercial value, and generally avoided rare species; especially those listed under the Endangered Species Act. Endangered species were excluded from compensatory mitigation consideration because they were too highly valued to risk their loss once listed under protection of the ESA. Negative impacts on endangered species were to be totally avoided in the first place. Similarly, if an entire ecosystem was very rare and composed of unique species, environmental impact analysis and mitigation would usually choose avoidance of negative impact over attempted compensatory mitigation. Compensatory mitigation was most often allowed by the regulatory agencies when the losses were economic (recreational, commercial fishing) rather than environmental (EQ).

The HSIs, HEP and IFIM generally worked well conceptually for "exact" compensation of fish or wildlife loss as long as the same measures were used to assess both the impact site, before it was impacted, and the compensatory habitat created or restored for mitigation. Loss of a large acreage with low average quality could be compensated by creating or restoring a small acreage with high average quality. The assumption was that, regardless of quality and quantity combinations, the *value* of habitat lost to water resource development was at least fully compensated by the *value* of restored or created compensatory habitat.

An important complication occurred when the consistent use of the same species index over impacted sites and compensation sites was impractical because the value of HUs varied among different species. Two or more species with the same HSI, or increment of change in HSI, usually differ widely in abundance, production, or other measures proportional to species value. In addition, human preferences for different species often vary depending on perceived utility and/or value. Even for endangered species, "charismatic megafauna" (e.g., bald eagle, salmon) are valued more highly than small and cryptic forms (e.g., freshwater mussels, snail darters). There appears to be no practical way that habitat units of different species can be made reliable indicators of relative value for comparative analysis.

Another problem arose when ecological settings for the compensation site and the impact site differed substantially. The interactions among habitat variables then became different and increased the probability that the same index represented different species abundance or other performance measure. For this reason on-site compensation was preferable to off-site compensation except when it was impractical. In addition, the farther off site the compensation occurred, the more it altered the supply of resource with respect to human demand. The same resource production could become less or more valuable as a consequence. (This is not a problem for species recognized nationally as important because of their vulnerability to extinction but having no overt utility, because no local interest has the advantage of proximity.)

As used in the past, single-species HEP often did not capture all of national interest associated with ecosystem services (NRC 1999). This problem is associated with the degree that the habitat of a single species indicated *all of the value* that needed compensation. For this reason many of the species selected for HSI development were dominant species of high recreational and/or commercial value. They captured much of the value in their index. In some cases, the species were selected as habitat indicators for a collective productivity of valued resources, such as an abundant forage species that sustains a number of more directly valued sport and commercial species. Even so, it was difficult to assure that all ecosystem services and values associated with an impacted site were captured in the habitat requirements of a single species. As a consequence, the habitat focus of compensation typically ignored effects on water storage, water treatment, storm-surge reduction, and other ecosystem services that could have been important.

Finally, there is the issue of predictive accuracy. Brooks (1997) has criticized insufficient verification for existing HSIs and there is some evidence that existing models have not proved as effective as once hoped. The more universal chronic complaint is about the lack of evidence for or against the continued use of an existing model. However, this general complain also applies to other ecosystem management models used by government agencies.

Many of these issues have proved to be problems for ecosystem restoration use. Another issue is the degree that a single species can inclusively indicate more natural conditions for the entire habitat and community complex comprising the ecosystem. The most influential attributes of a species' environment form a subset of all attributes affecting the community-habitat complex. The best indicator species are often dominant plants, for which few HSIs have been developed. However, community-habitat indices may be a better general alternative to indicator species for representing the relative naturalness of ecosystems.

Even so, the most socially *significant* resources of ecosystems are likely to be scarce species in many decision processes. HUs based on the needs of the scarce species could be useful, once developed, but few now exist. Because they are too narrowly focused to be inclusive indicators of a more natural *ecosystem* condition, the most effective planning use can be made of them when they are linked with a community-habitat model of relative naturalness and integrity. In that process, the degree of restoration applied to a more natural ecosystem condition can be evaluated against incremental cost and outputs from that model can serve as inputs to the single-species habitat models to evaluate the effect on the significant resource.

### 4.3.2 Community-based Habitat Indexes

Community habitat suitability models offer improvements over species level models for indicating the naturalness of ecosystems in a number of ways. The WCHE, RCHARC and IBI models, for example, are based in structural indicators of community naturalness. They anchor their maximum index value to a native species diversity or other native biodiversity measure existing in the most natural state determined from reference conditions. Because they indicate the relative effect of humans on communities they can be useful for formulating and evaluating according to the ecosystem restoration planning objective. However, unless the entire community is an ecological resource of national significance, or there is a known direct relationship between the community index and the condition of the significant resource, some other measure of resource significance is needed to evaluate the restoration of resources of significance.

An important limitation of community-habitat models of all kinds is that they are typically based on the habitat needs of the more common species in the ecosystem—that is, the species that are most readily investigated for model development. Figure 4.1 illustrates different possible relationships that might exist between the habitat suitability indices of vulnerable resource species, which probably qualify as resources of national significance, and a native species richness indicator generated by community-habitat model. If the vulnerable species and the native species follow the same patterns of relative abundance and rarity in the system, a relationship like B in Figure 4.1 would exist and could be used to guide restoration species viability as well as the full complement of species and functions. Even so, extrapolations of relationships to the rarest species is prone to uncertain results. In situations where the vulnerable species are very rare in the ecosystem and are likely (example C) to be restored to the community-habitat complex only as the ecosystem approaches a fully natural state, most of the vulnerable species will fall into an uncertain restoration status.

The broadly adapted species in ecosystems are often among the more common species that dominate the restoration of a disproportionate amount of the function other than that related to sustaining the most sensitive species. In relatively few ecosystems (some isolated western spring systems, for example), globally rare species dominate (example A). These are typically unique ecosystems, however, for which there are no generally applicable models. Development of a biodiversity model of relative naturalness for such conditions would include most of the globally rare species because they often dominate in these simple systems. In most ecosystems, however, there is little evidence that globally vulnerable species are consistently among the dominant species.

The relevance of the relationship that actually exists in restoration prospects is important for prioritizing restoration actions. If the primary justification for the proposed restoration is reducing species vulnerability and model B is correct, then a partial restoration action would contributes proportionally little to that end. Most or full restoration should be the objective to assure a significant fraction of the vulnerable species will recover. If the justification is based more on the recovery of services other than the genetic information in rare species, partial restoration may be more suitable. For example, restoration of erosion control and nutrient retention may occur relatively quickly as biomass accumulates in the restoration of a small fraction of the community. The stability of function is likely to increase with further restoration of community components, but not at a consistent rate like that indicated in model B.

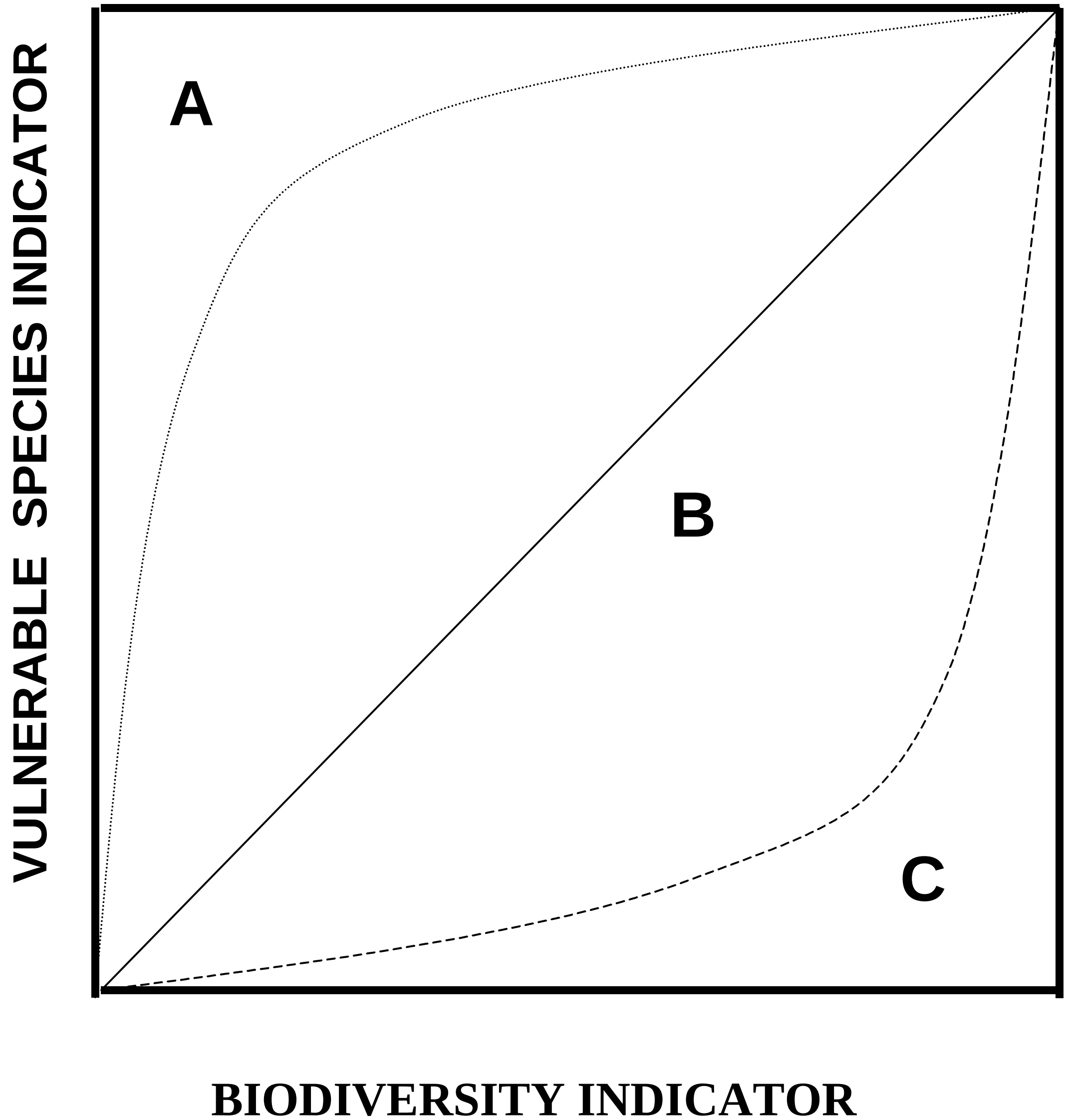


**Figure 4.1. Possible relationships between a theoretically inclusive habitat suitability indices for native species richness and for the vulnerable species in ecosystems, depending on whether the vulnerable species are more abundant (A), less abundant (C) or equally distributed in abundance (B). In most conditions, only a fraction of the more common species are included in the development and calibration of such models (as indicated by D) because of sampling limitations. See the text for discussion of the Relationships.**

The same caveats hold true for the HGM approach, which assesses the naturalness of ecosystem function based on a suite of functional capacity indexes calibrated against a gradient of human effect, with the most natural state of a carefully classified wetland type having the maximum index value. But because functional capacity often recovers quickly with the restoration of the more common species (as discussed in Section 3) scarce resources may not be recovered in anything short of full restoration. Even then, this and other ecosystem-level models tend to overlook important connections to the larger landscape.

Similar to single-species habitat indexes. community-habitat or ecosystem index models are typically limited to a local planning perspective. They tend to externalize large-scale landscape features and processes that can be very important for assuring the natural processes of importance to significant resources are restored. RCHARC, for example, indicates variation from a natural state based on the velocity and depth alterations, but does not address other factors that might impede recolonization off site, once desired conditions are created or restored. This requires an alternative, usually ad-hoc (professional judgment), evaluation of landscape level influences.

None of these index models forecast or evaluate the change in the national resource condition, which would be challenging, but theoretically possible, The index might include weights proportional to local and national contribution to the resources of significance, such as relative abundance or geographical area. This type of information is invaluable for evaluating the significance of plan effects. It is complicated however by the fact that indexes nay not reflect differences in habitat-community productivity very well. Some do not have any production factor and others simply average production factors (relative abundance measures) in with other factors. The optimum conditions determined for resources of significance in two different ecosystem areas might produce an order of magnitude difference in the production of individual organisms contributing to the national resource. These differences may or may not be integrated into the calibration of the models. Existing indexes need to be reconsidered in a national perspective to assure that they account for such differences.

When the resources of significance are single species, a single species-habitat model, or an array of such models, can be checked against the proposed state of naturalness indicated by a community-habitat model to determine how the relative performance of the resource is likely to respond to the more natural condition with respect to its optimum. Like other models applied only in a local planning context, if there is not a clear idea increased abundance, reproduction rate, or other measure of resource improvement, there will be no clear idea of how that improvement relates to the state of the resource nationally.

The predictive capability of both natural integrity and specific resource response to restoration measures decreases sharply as ecosystems become more generally modified and undergo more intense stress. Existing models are based on the assumption that the traditional concept of resilience is in effect and that processes of natural colonization and succession are consistent with that concept. As the probability that ecosystem response will “flip” into a new stability regime increases because of widespread disturbance, existing models become less reliable for formulation and evaluation. Two strategies for controlling this source of risk include 1) emphasizing restoration where the traditional “rules” of resilience are most likely (the short term strategy which avoids widely disturbed areas of ecosystems), and 2) developing more spatially explicit and comprehensive models based on improved understanding of culturally fragmented and stressed ecosystems.

Brief descriptions follow for nationally recognized community-habitat index and ecosystem functional capacity index models:

**Wetland Valuation Assessment (WVA).**. This is an interagency product developed for use in coastal wetlands, mostly in Louisiana, to carry out authorities under the Coastal Wetlands Planning, Protection, and Restoration Act of 1990 (Louisiana Coastal Wetlands Conservation and Restoration Task Force 1991, Environmental Work Group 1998). It is a community-level HSI/HEP approach. The wetland types include freshwater marsh, brackish marsh, saline marsh and cypress-tupelo swamp. Just as for species HEP, expediency in carrying out the federal law was an important criterion for assembling the community HSI models. The WVA is the only community model described here that does not establish its maximum index value based on some undisturbed natural state. The maximum habitat suitability is based in a concept of some community-level "optimum" based on an "average" optimum condition determined from the HSIs of 31 high-profile indicator species. It is, therefore, not an ecosystem restoration model in the narrowly defined sense.

**Index of Biotic Integrity (IBI)** . The Clean Water Act set as its objective, the restoration of physical, chemical and biological integrity of the Nation's waters—i.e., restoration of aquatic ecosystem integrity. Recognizing that biological integrity depends on suitable physical and chemical conditions, Karr (1981, 1986) devised an Index of Biotic Integrity (IBI) to assess progress in meeting an ecosystem restoration objective, such as might follow from the elimination of a chemical pollutant. The IBI is a multidimensional index of different community habitat conditions summed from a suite of subordinate indexes based on the richness, composition, and health of representative members of a community group (Karr 1991). It has been developed for fish and invertebrates. Of the indexes described here, the IBI is the only one described among ecological indicators for the Nation by the NRC (2000).

The IBI is based on the regional native biota indicative of unique communities and is anchored in the community and habitat integrity of undisturbed ecosystems. It has been most thoroughly developed for Midwestern streams, but is undergoing development and evaluation in wetlands (Minns et al 1994, Burton et al. 1999) and other stream ecosystems (e.g., Simon 1999). The Midwestern fish IBI is composed of 12 subordinate indexes, each of which is ranked 1, 3 or 5 indicating the variance of a community from the unimpaired natural community condition. The best score is 60 points and the lowest is 12 points. The IBI has stimulated widespread interest in applications elsewhere in recent years. Plafkin et al. (1989) described rapid assessment protocols using an IBI approach and discussed the potential for guiding restoration. Because it is designed specifically to restore more natural ecosystem integrity, the IBI leads among models for guiding the restoration of more natural ecosystem conditions.

**Wildlife Community Habitat Evaluation (WCHE).** The Clean Water Act authorized the Corps to regulate discharge of dredge and fill material into the Nation's waters with the intent of mitigating impacts where practicable. Wetlands have received exceptional attention due to state interest and U. S. executive-branch policy. Numerous wetland evaluation methods had been developed (Bartoldus 1997) but none satisfied Corps regulatory needs. Schroeder and Haire (1993) had reviewed existing community-level habitat indices in response to a need expressed by the U. S. Fish and Wildlife Service for practical assessment tools more comprehensive in scope than the single- or multi-species HSI models existing at the time (e.g., FWS 1981, Short 1984, Adamus 1987). Out of that philosophy Schroeder developed a small series of upland models and won the attention of the Corps who funded development of a WCHE for forested wetlands in Maryland (Schroeder 1996a and b).

The forested wetland WCHE developed community-habitat suitability indices for community assemblages of native species based on the relationship of native vertebrate species richness to several habitat variables including habitat edge and isolation attributes (Schroeder (1996a and b). The native vertebrate species richness is the criterion used to gage community response to habitat suitability . A maximum suitability is indicated for the condition that supports the maximum number of forest-interior native species. An important conceptual advance was incorporation of landscape-level habitat features that reflect the effect of habitat fragmentation. However, a disadvantage in the single wetland model so far developed is the lack of hydroregime habitat variables that might link vegetation form and other ecosystem attributes to Corps restoration measures. Because it is based on a scale of relative naturalness, this model has potential for utility in place of or in addition to the IBI and other community-habitat index models.

**Riverine Community Habitat Assessment and Restoration Concept (RCHARC).** The Corps also has invested in the development of a model for use in environmental mitigation of physical impacts on flow regimes in large rivers and for guiding river-ecosystem restoration decisions (Nesler et al. 1995). RCHARC derives its underlying concept from single-species HSIs developed for IFIM . It relies on the relationship between most fish species contributing to the membership of the river community and the distribution of flow velocities. Unlike the WCHE for forested wetlands, RCHARC is linked to hydro-regime management.

RCHARC was developed and used for the Missouri River and has had limited application elsewhere (e.g., Apalachicola system). Like other habitat-based relative indices of community condition, the maximum index value is anchored in that habitat condition resulting in maximum species richness observed in a range of flow conditions. Being narrowly defined in terms of flow dynamics, RCHARC as it is presently configured predicts habitat suitability only for flow dynamics. The model cannot predict accurately for a site where other variables are limiting, such as oxygen or temperature. Like other community-habitat models that attempt to characterize a range of relative naturalness in ecosystem condition, this model is suitable for restoration purposes, but would be more suitable if other habitat variables were included in addition to hydrologic variation. Also, in the highly modified, large river conditions for which it was developed, it is difficult to separate natural variation from variation caused by human impacts.

**An Ecosystem Functional Capacity Index—The Hydrogeomorphic Approach.** Only one method develops indexes of ecosystem function. Following an executive order for no-net-loss of wetland function and value in 1990, a technique was sought to assess wetland ecosystem functional capacity. The Hydrogeomorphic (HGM) Approach was developed in prototype by Brinson (1993) and its development is continuing. With Corps funding, Smith et al. (1995) expanded the concept and initiated development of specific models for different wetland types. The basic premise made for calibrating HGM models is that unimpaired ecosystems within each ecosystem type are fully functional (1.0) and human alteration reduces the functional capacity index (FCI) along a scale between 1.0 and 0. A wetland classification has been completed to determine the fully functional benchmark ecosystems and a number of type models have been completed. Wetland types are defined by hydrologic, climatologic and geomorphologic settings and associated communities (Brinson 1993). While theoretically applicable in any ecosystem type, the method has so far been applied only to wetlands.

Somewhat like the IBI concept, the HGM Approach uses a number of functional capacity indices to define the ecosystem condition. These vary in type and algorithm depending on the wetland type. Unlike the IBI, however, the FCIs were not intended to be summed, averaged, or otherwise integrated into a single index value. Wetland functional attributes depend on wetland type and cannot be compared directly across wetland types. While some types of functions are held in common among all wetland types, such as water storage and habitat functions, many functions are limited to a subset of wetland types. Organic detritus export, for example, is a function only of wetlands occupying open basins. Each function is described by its own functional capacity index, which is calculated by an equation assembled from a number of indicative community habitat variables (e.g., suspended solids and water level fluctuation).

The HGM Approach has potential for use in guiding wetland and other ecosystem restoration actions. It has one important advantage over the community HSI models in that it is more inclusive of all ecosystem functions relevant to ecosystem services. King et al. (2000) are studying the possibilities for a weighting method to create a wetland value index from functional capacity indices based on ecological context, social context and human preferences. The HGM Approach, however, retains most of the shortcomings of any relative index model. The predicted results have little meaning outside the ecosystem reference framework. Different ecosystems can only be compared through the functions they hold in common. In addition, the indices to the different functions do not directly reflect the biodiversity variables that appear to influence functional stability in support of service reliability. Even so, the HGM Approach characterizes the relative naturalness of ecosystems through their important functions and can be useful for evaluating measures taken to achieve the ecosystem restoration planning objective.

### 4.3.3 Ecosystem Process Simulation Models

Models that simulate ecosystem function and structure are based in concepts dating back to Lindeman (1942) and Odum (1957). They are variously known as process models, simulation models, compartment models, input-output models, mechanistic models, modular models, and dynamic state models. Their common intent, however, is to simulate natural process rates and output amounts as closely as needed for the model purpose. They are typically developed from theoretical mathematical descriptors of process and form but may be hybrid models including both theoretical and empirical elements (statistical equations). Many such models have been developed for research purposes, such as formulating a hypothesis of how complex ecological mechanisms might interact to generate an ecosystem output, which is then compared to real-world observations. Fewer process simulation models have been developed and widely used for management applications because they usually take more time to develop than allowed by statute mandates. They frequently require local calibration with extensive date, are relatively costly to use, and often involve a disconcerting array of variables and outputs for practitioners typically focused on one or two model outputs. NRC (2000) refers to a number of qualitative concept models and related quantitative models of ecological process relevant to development of national ecological indicators.

The multitude of possible outputs and comparisons also can be advantageous for analysis of complex ecosystem process. Unlike index models, process simulation models can provide great flexibility in use and can enable direct comparison of numerous interactive outputs in response to inputs of simulated environmental stress or management change. Among the more useful capabilities is for analysis of management tradeoffs among ecological outputs in a “shared vision” approach to planning. In addition, the outputs from one model can be coupled to the inputs of other models in time steps that allow simulation of natural feedback effects and interactions among different modeled functions and structures.

Community-level structural and functional output from one component (e.g., vegetation form and production) can provide controlling inputs to species groups and to individual population components. Any number of significant output modules can be modeled at the species or ecological guild level. It is conceptually possible, therefore, to include both ecosystem-level measures of naturalness in model form and function and subsystem models representing resources of significance, and even feedback interactions between the two if appropriate. Hybrids of species-habitat index models and process simulation models have also been constructed (DeAngelis et al.1998), but feedbacks from the index models are conceptually difficult.

At the model core are state-variable equations that quantify a condition at a particular time, but vary through time as model inputs vary. A common state variable condition is biomass density (e.g., kg/hectare) of a functional community group, such as primary producers or herbivore secondary producers. The state variables change as input conditions change with each time step included in the model. Time steps vary greatly, from minutes to years depending on the scale of interest and data availability. The state variables form compartments with driving inputs and outputs that serve as inputs for other compartments. The state variables are linked by equations defining relationships with coefficients influenced by other variables. Density-dependent feedback relationships are common in ecosystems and in process models. The amount of change in a state variable often determines in part the amount an influential variable changes. Food-web feed backs combine with habitat variables to determine the functional stability of state variables.

The basic input variables used in aquatic ecosystem simulation models typically include initial biomass of producer groups, the driving energy input (usually solar and biochemical), controlling nutrient concentrations, water flow, topography (channel and basin form), temperature and other environmental-constraint data. Temporal variation in solar energy and water discharge are necessary inputs for fully simulating ecosystem dynamics. Depending on purposes, simulation models may either explicitly or implicitly cycle nutrients and track other material flows.

**Spatially Constant Models**. An early example of an aquatic process model is Clean-X developed for open waters of lakes (Scavia et al. 1974) and a stream model by McIntire and Colby (1978). The most important conceptual model for streams, the River Continuum Concept A more recent aquatic ecosystem simulation model, CASM (Comprehensive Aquatic System Model), has been used to assess ecosystem structural and functional relationships (DeAngelis et al. 1989) and risks of dysfunction from contaminants and other stressors (Bartell et al. 1999). Friend et al. (1997) described a process-based, terrestrial biosphere model of ecosystem dynamics (Hybrid v3.0) for global assessment. This is a general application model of carbon, water and nutrient cycles coupled with soil, plant and atmospheric systems. Models of this scale may have potential for analyzing cumulative effect of restoration process to regional or global process. The Corps has invested in a Successional Dynamics Simulation (SDS) model for upland terrestrial conditions affected by military operations (McLendon et al 1998).

**Spatially Variable Models**. Spatially explicit process models are relatively recent additions to simulation model advances. Their development has been closely coupled with Geographical Information Systems (GIS). Especially targeted for modeling attention have been the movements of living organisms through landforms and across landform boundaries (the so called Mobile Animal Models [MAP] described by Dunning et al 1995. Rudimentary spatially explicit community models have been developed, such as the wetland model described by Poiani and Johnson (1993). One of the more elaborate examples of spatially explicit models is ATLSS (Across Trophic Level System Simulatino), which has been developed for South Florida study of Everglades restoration (DeAngelis et al. 1998)

Recently two spatially explicit models have been developed with potential for aiding restoration process: FRAGSTATS ( McGargigal and Marks 1995) and PATCH (Schumacker 1998). FRAGSTATS provides the user access to a number of algorithms for calculating landscape-scale metrics such as habitat area, patch sizes, patch pattern, and total edge development. FRAGSTATS has been used to assess landscape suitability for both single species and groupings of wildlife (Rosberry and Sudkamp (1998), Glennon and Porter (1999), Penhollow and Stauffer (2000). PATCH provides a GIS-based platform for tracking wildlife populations through time and space While PATCH will track several populations in a landscape context simultaneously, it does not account for population interactions. FRAGSTATS and PATCH offer an advantage over HSI in their potential for evaluating the importance of habitat connectedness to other habitats for restored habitat colonization from dispersing populations. To the extent they are most useful mostly for simulating environments of individual populations, they have some of the same limitations as species-based habitat suitability indices.

A Geographical Information System (GIS) is the usual means for organizing and overlaying data in a map-like or geographical format. GIS is not a model, but a database management system that is increasingly integrated with ecological models. A GIS may be used to input information, house model process, and output information in map form. GIS software has greatly facilitated model use and development for spatially explicit natural resource inventory and management. A common use of GIS is to store ecological data on land form, vegetation, land use, species and other distributions according to map coordinates. A national-scale example of this use is the development of GAP Analysis (Scott et al. 1993) by the U.S.G.S. For GAP analysis, vegetation, species distributions and property ownership boundaries are overlain to assess the species distributions with the intent of identifying key areas of high biodiversity and high vulnerability based on potential land and water use. All of the United States is expected to be completed over the next few years. GIS also is widely used to organize information at much smaller geographical scales. The upper Mississippi Corps districts, for example, use it to carry out the Upper Mississippi Environmental Management Program and interfaces it with a simple process simulation model that predicts plant succession to forecast habitat condition changes. A good example of GIS use in a process simulation model is ATLSS, which is used to analyze plans for restoration of the Everglades and adjacent ecosystem conditions in South Florida (DeAngelis et al.1998).

## 4.4 Choosing Models for Restoration Planning

### 4.4.1 Importance of the Systems Context

Determining the "best models" to use for guiding restoration of more natural ecosystem conditions and associated resources of significance is situational, depending on the complexity of the natural state and the alterations that have occurred. Just about any rigorously applied model type, including physical models, may suffice for situations where there has been very little ecosystem change from the natural state, the condition to be restored is closely connected to the restoration site, full restoration is feasible (at least to the level indicated by an existing natural reference condition), and the source of the deficiency in resources of significance is easily identified and removed. However, most models do not explicitly evaluate sustainability, but rather assume that a close relationship exists between the indexed performance measure and sustainability. Such assumptions are unevenly justified.

A model guiding restoration to a fully natural biodiversity based on existing reference conditions, including some idea of the abundance of significant resources, involves the least risk that resources of significance will fail to be restored as forecast as long as the significant resources are also found in the natural reference conditions. They are also most likely to restore a sustainable state, if the existing natural state is sustainable. However, the influential landscape variables that often determine local sustainability are frequently not addressed in most existing models, only a few process simulation models approach this level of comprehensiveness (e.g., DeAngelis et al. 1998). For example, major changes in precipitation, air temperature, cloud cover, and sea level could greatly modify, even eliminate many existing ecosystems over a period of 50 years. Models incorporating more than one measure (multicriteria models) of biodiversity/ integrity are more likely to inclusively represent natural biodiversity than most single-species models. While single species models can be useful when chosen with the entire ecosystem in mind, or (sometimes) as indicators of resource significance, they are easily misused.

Many restoration proposals target partial restoration of naturalness under more complicated conditions involving much fragmentation of the original ecosystem conditions and many different sources of stress and pathways to altered states. They involve systems with many natural and human influences, interactive feedbacks, landscape-scale considerations, remotely located and subliminal limiting factors, and other complex interactions, such as occur in many culturally modified parts of ecosystems. As conditions grow more complicated, the advantages of spatially explicit process simulation models begin to outweigh the accessibility and low-cost advantages of other models. Regardless of model choice, when partial restoration of ecosystems is under consideration, the relationship of output indicators for resources of significance and output indicators of naturalness need to be defined clearly to assure consistency with Corps restoration policy.

### 4.4.2 Modeling For Common and Scarce Biodiversity

The previous review of ecological principles in Section 3 suggests that some multi-dimensional measure of natural biodiversity may hold promise as an indicator for most, if not all, of the non-monetary benefit sustained by fully natural ecosystems. Several types of “biodiversity” models characterize relationships between habitat inputs and community or ecosystem outputs along a gradient of human effect anchored in the most natural condition. However habitat and community measures of biodiversity in most existing model types are most reliable for the more common ecosystems components and aggregate function and structure. They often loose predictive reliability for the scarcest components, most likely to qualify as resources of significance, such as the globally rare species, as indicated in Figure 3.5 and Figure 4.1. This deficiency has to do with the practical problems associated with calibrating models, which typically are based on the more common components of ecosystems.

Even species diversity measures frequently miss explicit inclusion of the globally rare species, which often qualify as the resources of greatest national significance. Thus the habitat-community relationships defined for the more common species must be assumed to hold for the rarest species as well. This assumption becomes increasingly secure as more the ecosystem needs of more of the species the community are included in the model. Even so, the uncertainty associated with inherent variation, often determined by random events, increases as the restoration justification increasingly hinges on the response of a very few species to restoration measures. Few commonly used models have addressed this uncertainty issue adequately.

When the resources of significance are based on scarce biodiversity, as indicated by the uniqueness and vulnerability of species, model selection depends on how many of the species in a community qualify as significant resources. In a situation where only one or two species qualify as scarce in a restored community, it would typically be best to use a community model to guide the restoration of the natural support system, and species-habitat models to check on whether or not the suitability of habitat has been obtained for the significant resource species. As more species in a community qualify for scarcity status, the added benefit of coupling with individual species-habitat models decreases. For greatest utility, the coupling of community and species models requires that all of the input conditions (habitat variables) for the species-habitat model also would be included in the community-habitat model. This will require a coordinated effort that has yet to be done. Thus, the most useful models for recovering overly scarce biodiversity have yet to be developed, either in index form, the more elaborate form of process simulation models, or in hybrid models.

Existing methods and models can be usefully applied to formulate and evaluate for scarce biodiversity resources, but with heavy reliance on professional judgment and concept models of the system context. Concept models should be developed with special attention to the risks and their management. Once species, guilds or entire communities have been determined to qualify as resources of significance, the primary challenge is to identify the risk of project failure in realizing their recovery and managing that risk to an acceptable level. As a general rule, risks are lower when the project area to be restored is immediately adjacent to and functionally closely tied to a large, fully natural area that supports thriving remnants of scarce resources, and when the causes of degradation are few and easily corrected. As the project area becomes more disconnected from the naturally intact ecosystem and the causes of degradation become more numerous and complex the risk of realizing a sustainable contribution to NER increases.

The existing set of modeling tools are more reliable for restoration plan forecasting when the resources of significance are determined to be associated with restoration of the more common biodiversity in ecosystems—such as the production and biomass functions that contribute substantially to aesthetic, recreational, flood damage reduction, water supply, and water quality services. The tools are more dependable because the resources are abundant enough to have been well studied, in contrast with the scarce resources. However, if resource scarcity is the most important determinant of NER qualification, substantial improvement of existing models and methods is in order.

### 4.4.3 Existing Model Limitations

Few existing models can be used without extreme care and understanding of the underlying project ecosystem condition and its systems context. While species-based HSI models are numerous, easy to use, and immediately available, and are relatively inexpensive (Figure4.2), they rarely capture all of the important habitat/ ecosystem elements to assure a more natural, self-regulating condition will result, or all of the justifying value needed to restore ecosystems. Species-based HSIs are not scaled based on ecosystem integrity and can only be used to indicate a more naturally integrated ecosystem condition if the HSI value is known for the targeted restored condition. Few existing single-species HSI models satisfy these criteria well, but ecosystems might be characterized by new models for native dominant and keystone species, including dominant plant species, scaled against a gradient of altered conditions anchored in the most natural ecosystems. Several species HSIs might be used to "bracket" the community-habitat relationships satisfactorily, but the need for many new models and much calibration offsets the main existing advantage of HSI models. In addition, few HSI models now exist for the most vulnerable species or guilds in aquatic ecosystems, and would need development for use either alone or with models of ecosystem naturalness.

Community HSIs indicate relative ecosystem naturalness and associated non-monetary benefit more inclusively than species-based models because they link habitat more broadly to ecosystem components or functions. Among existing models, WVA appears to have many of the same limitations of the species HSIs from which it was derived. It is based on the optimum needs of relatively common species; not on a scale of relative naturalness or on scarce resources of environmental significance. The HGM approach l links directly to the naturalness of ecosystem functions through FCIs, but, like all index models, they cannot be readily compared across local ecosystem conditions to aid in restoration priority decisions.

Sustainability of ecosystem function and structure is an increasingly important criterion for model selection, and the closely related concept of self-regulation is a defining attribute of more natural conditions in Corps policy. However, concepts of ecosystem health and cultural integrity suggest that sustainable states can coexist with substantial human alteration in carefully considered situations. Principles of forest, range, and other natural resource management have assumed such for many decades, sustainable management being a cornerstone of wise resource use. Index models do not address functional stability, self-regulation, and sustainability of ecosystem structure explicitly, however. Species HSI models usually provide little theoretical or practical insight into the sustainability of the conditions they indicate. While it might be assumed that the FCIs of HGM, or the HSIs of communities are proportional to an ecosystem's capacity for self-regulation, functional stability, and sustainability of structure, these attributes of ecosystems have not been examined critically. Because all of these models focus on local conditions, they fail to capture all of the landscape attributes of the entire ecosystem that are so important in determining sustainability of scarce ecosystem structure.

Models vary in the extent to which they have been developed. By far the greatest number if models available "on the shelf" are single-species HEP/HSI models. But few existing models appear suitable for environmental resource evaluation. The ecosystem index models that have the greatest potential for use in a wide variety of ecosystem types are the IBI, FCI of HGM, and WCHE, but none have been developed for a full range of ecosystem conditions of interest to the Corps. The IBI has the longest history and diversity of development, but even among stream ecosystems for which it is best developed, many stream ecosystems remain to be calibrated. HGM has yet to cover all wetlands let alone all other ecosystems of interest to the Corps. The WCHE is most limited in this regard, having been developed only for one type of forested wetland and several upland ecosystems. Any of the ecosystem index models would require considerable investment to cover the variety of ecosystems managed by the Corps, but IBI and HGM have had the greatest investment so far. Integration of IBI, HGM, WCHE and other index model attributes is a possibility that ought to be considered as well, if an index approach is to be emphasized in the future.

Ecosystem index models also make broad assumptions about the “tightness” of relationship between selected indicator species and the entire ecosystem. Most models disproportionately rely on fish, invertebrate, or bird subsets of the community-habitat relationship to represent the entire ecosystem condition. The taxonomic groups chosen for characterizing integrity may not characterize to fine enough degree all of the relevant attributes of a more natural condition, nor the habitat needs of the scarce resources of significance. Complete methods would need to account for this potential deficiency by assuring the biodiversity measure in the index is inclusive of the significant resources or by including a separate relationship between vulnerable-species and habitat conditions.

Many of the shortcomings of index models are addressed in process simulation models, which ultimately offer the greatest flexibility in use and the greatest management insight with respect to the output generated with incremental additions of restoration measures. Self-regulating mechanisms are built into such models through density dependent and other feedback relationships. Functional stability and sustainability can be analyzed directly from the dynamics of modeled output, but still remain among the more difficult attributes to model. Functional and structural changes can be examined in explicit estimates of actual output amounts and in spatially explicit dimensions. The effects of uncertainty can be assessed through analysis of the sensitivity of output to the uncertainty associated with specific model structure. Process simulation models are more typically “theoretically rigorous” because process understanding is an important objective. Because of this, they are among the best models for organizing information adaptively through time as new information becomes available. In terms of basic ecosystem structure, processes, and interactions, similar principles operate across all ecosystems to which such models apply.

However, process models can be “information hungry” and more time consuming, especially when precise prediction is a high priority. Much can be learned about how ecosystems work during assembly of process models, but the ultimate models for evaluating nonmonetized environmental benefits are years away even if research investment were immediately and substantially increased . The objections to process models expressed two decades ago (leading to the emphasis on index models), having to do with inadequate portability and computational capability, have been greatly diminished by the widespread availability of powerful personal computers. Even so, the details of resource partitioning into communities of different species richness and functional stability requires much research and development. In the process of assembling such models, much more could be learned than from index models about managing ecosystem process for more reliable service delivery across all natural and enhanced services. Process simulation shows the most promise for incorporating tradeoff analysis within single model operations.

The existing state of ecological knowledge, management need, and model development capability leads to the conclusion that, in the near term, a selection of environmental-benefits estimation models needs to be made available for resource management planners. If recovery of resources of significance and ecosystem naturalness are to be jointly considered objectives of ecosystem restoration, the most useful planning models will be capable of representing the responses of both to restoration measures in naturally variable settings. In the short-term, this may require a combination of a suitable community HSIs or ecosystem FCIs and species or guild HSIs. Much development is needed, however, because many ecosystems and resources are not now addressed by existing models. In the longer term, greater development and use of process models ought to be considered because of their more explicit estimation of actual output amounts, their capacity for organizing great amounts of model input and process information into simultaneous forecasts of numerous and diverse outputs, and their long term adaptability to management needs.

## Section 5. Corps Standards for Plan Evaluation, Comparison and Selection

This section reviews the standard Corps planning framework used for the evaluation, comparison and selection of project plans formulated to serve traditional Civil Works purposes, and how it has been adapted to the ecosystem restoration purpose. As used here, the term "plan evaluation" refers to the quantitative measurement of an alternative plan's negative and positive effects. Plan comparison refers to the analytical procedures used for examining the economic efficiency implications of and tradeoffs among alternative plans, and plan selection standards refer to rules for justifying plans for funding.

### 5.1 Overview of Policy Standards for Single and Multiple Purpose Projects

Corps planning standards for evaluating plan benefits, and for comparing and selecting among formulated alternatives in the case of traditional "National Economic Development" (NED) projects, "National Ecosystem Restoration" (NER) projects, and multipurpose NED/NER projects are summarized in Table 5.1 and reviewed below.

**Table 5.1 Corps Planning Standards for NED & NER Purposes***

| | **Plan Benefits Measure** | **Plan Comparison Procedures** | **Plan Selection Rules** |
|---|---|---|---|
| **Single Purpose NED Projects** | "Contributions to national economic development (NED outputs) are increases in the net value of goods and services, expressed in monetary units." | Benefit-cost analysis: monetary NED benefits less monetary NED costs | "For all project purposes except ecosystem restoration, the alternative plan that reasonably maximizes net economic benefits consistent with protecting the Nation's environment, the NED plan, shall be selected." |
| **Single Purpose NER Projects** | "Single purpose ecosystem restoration plans shall be formulated and evaluated in terms of their net contributions to increases in ecosystem value (NER outputs) expressed in non-monetary units." | Cost effectiveness and incremental cost analyses based on non-monetary NER benefits and costs to implement plans | "For ecosystem restoration projects, a plan that reasonably maximizes ecosystem restoration benefits compared to costs, consistent with the Federal objective, shall be selected. This selected plan must be shown to be cost-effective and justified to achieve the desired level of output. This plan shall be identified as the NER Plan." |
| **Multiple Purpose NED/NER Projects** | Multipurpose plans are to be evaluated in terms of both (monetary) NED outputs and (non-monetary) NER outputs | "Recommendations for multipurpose projects will be based on a combination of NED benefit-cost analysis, and NER benefits analysis, including cost-effectiveness and incremental cost analysis." | "Projects which produce both NED benefits and NER benefits will result in a best recommended plan so that no alternative plan or scale has a higher excess of NED benefits plus NER benefits over total project costs. This plan shall attempt to maximize the sum of NED and NER benefits, and to offer the best balance between the two objectives" |

Source: Chapter 2 of the *Planning Guidance Notebook* (ER 1105-2-100; April 22, 2000).

## 5.2 Economic Development Projects

The planning standards used by the Corps for project planning in the case of traditional Civil Works purposes are documented in the so-called *Principles and Guidelines* (P&G) as interpreted by Corps regulations set out in the *Planning Guidance Notebook* (PGN).[3] These define the overall Civil Works objective as the contribution to national economic development (NED), and require the Corps to estimate the NED costs and benefits of alternative project plans. NED benefits are defined as the economic value, expressed in monetary terms, of increases in the national output of goods and services as measured by users' aggregate willingness-to-pay (WTP) for additional units of services produced by a project plan. Aggregate WTP for a change in some service reflects the economic value of that change, as measured in terms of each affected individual's own assessment of his or her utility (i.e., based on individual preferences).

While not universally recognized, the NED concept of service benefit encompasses the economic value of all ecosystem services gained or lost by a project plan, including those services that are most closely aligned with the natural parts and processes of ecosystems (Shabman, 1993). However, because the ways in which these "natural" services contribute to human welfare often can not be readily traced and valued in monetary terms, Corps rules require that project plan effects on significant ecosystem attributes to be measured in physical/biological terms and recorded in the "Environmental Quality" (EQ) account.[4]

At the same time, however, Corps regulations establish a decision rule for plan selection that gives primary consideration to the NED (monetary) effects of plans. The PGN says that the recommended plan for Federal action in any NED project context is to be the alternative plan with the greatest positive net NED benefits (i.e., excess of money benefits over costs) that is consistent with environmental protection. In other words, the rules impose a "national economic efficiency" standard for plan selection, subject to environmental constraints set by established law and regulation. As discussed in more detail in Section 6, the conclusion that a water resource project that generates positive net NED benefits is in the national interest is based on the "potential compensation principle". This says that if those individuals who gain from a project could fully compensate those individuals who lose and still be better off themselves, then the project would increase overall national welfare.

## 5.3 Ecosystem Restoration Projects

Corps planning regulations establishes different plan evaluation, comparison and selection standards for project plans formulated to serve the NER purpose. Unlike traditional purposes, Corps rules do not require the monetary valuation of NER outputs produced by plan alternatives, or the use of cost-benefit analysis to identify and rank economically efficient plans.

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[3] ER 1105-2-100; April 22, 2000.

[4] The P&G framework includes four separate accounts for evaluating and displaying the effects of alternative plans: (1) the NED account, (2) the environmental quality (EQ) account, (3) the regional economic development (RED) account, and (4) the other social effects (OSE) account. Only the NED account and EQ account are required for project evaluation, however.

Instead, Corps rules say that NER outputs are to be quantified in non-monetary units, and NER project plans evaluated using cost-effectiveness (CE) analysis to ensure that the least cost alternative plan is identified for any possible level of NER output. CE analysis weighs the costs of each project plan against its non-monetary measure of NER output. The CE analysis screens out plans that are not cost effective from further consideration to ensure that the least cost alternative plan is identified for each possible level of NER output. Any particular plan is not cost effective if the same or a larger output level could be produced by another plan at less cost, or if a larger output level could be produced by another plan at the same cost.  The plans that remain after this screening process is performed define the "CE frontier", or the set of cost-effective (or "non-dominated) plans associated with successively higher possible levels of ecosystem outputs.

Once all cost-effective plans have been identified, then "incremental cost" (IC) analysis can be used to help answer the question "What level of restoration output is worth it? The IC analysis identifies the incremental cost per unit output gained from moving from one plan to the next higher-output plan. This incremental cost and value information helps to identify plans that capture production efficiencies with respect to NER output along different segments of the CE frontier (i.e., output ranges).  Figure 5.1 illustrates the results of a simple example of cost-effectiveness and incremental cost analyses for evaluating alternative restoration plans. Decision support systems have been developed (IWRPLAN, 1999) that make this type of analysis routine within the Corps. Such analyses can be implemented using any single metric of ecological output.

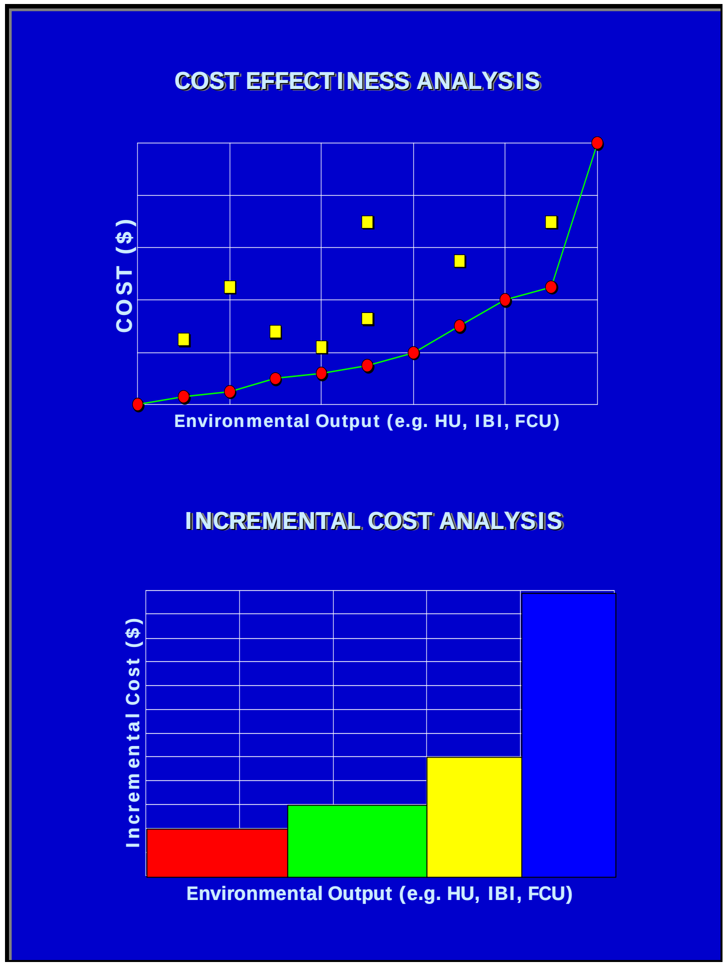



**Figure 5.1 Examples of CE/IC Analysis**

The CE/IC framework is applicable when NER outputs can be adequately characterized in terms of a single non-monetary variable. But in some restoration contexts it might not be reasonable or possible to adequately characterize and measure NER outputs in terms of one single metric. Consider a restoration project concerned with the protection of two endangered species that have substantially different habitat needs. In this case the contribution of any alternative plan to these objectives would likely require separate measures of NER output for each species of concern.

Cost effectiveness analysis is not applicable to the case in which NER outputs are measured in terms of two or more non-commensurate metrics. But that two-dimensional plan comparison framework can be readily extended to one defined over multiple dimensions. That is, a multiple criteria efficiency frontier (or “envelope”) can be estimated over three or more non-commensurate measures of plan effects. As with the basic CE frontier, the multiple criteria frontier defines the set of efficient, or non-dominated, plans. Consider a frontier defined over two NER outputs and plan implementation costs. In this case, the frontier identifies alternative plans for which more of one NER output could not be obtained through choice of an alternative plan without incurring higher implementation costs or obtaining less of the other NER output.

An analysis that traces out an efficiency frontier over multiple objectives can be very useful for informing decisions. However, more alternative plans will generally be identified as non-dominated as the number of plan effects considered increases; thus, fewer plans will be weeded-out as inferior. In addition, incremental cost analysis is not a particularly useful tool for informing the “is it worth it” question when non-dominated plans are defined with respect to multiple, non-commensurate criteria.

Efficiency analysis serves to narrow and illustrate tradeoffs among the set of plans considered for selection. Corps policy says that single-purpose NER project plans can be selected based on a subjective determination that non-monetary benefits are worth monetary costs, provided that the selected plan is shown to be cost-effective and NER outputs are shown to be “significant” based on institutional, public and/or technical recognition of importance. The significance test can be viewed as a way to document general demand for project outputs in the absence of monetary values providing a direct indication of demand. Other applicable project evaluation criteria relate to *effectiveness, acceptability*, *efficiency*, and *completeness*.

Figure 5.2 uses hypothetical project examples to contrast how the overall P&G framework is used for the traditional NED purposes, and how it has been adapted to the NER purpose. The project examples include a single-purpose NED project and a single purpose NER project. The second through fourth boxes moving down the center of the figure indicate what is measured by each of three successive project analysis steps. *Ecosystem outputs* represent the changes in ecosystem structure and functions expected to result from project activities. *Ecosystem service outcomes* represent changes in ecosystem services expected to result from changes in ecosystem structure and functions. Finally, *human uses and benefits* relate to monetary measures of the contribution to human welfare provided by project service outcomes. The numbered arrows that connect the first four boxes represent the various linkages among project activities, ecosystem outputs, service outcomes, and human benefits that must be estimated for comprehensive evaluation of plan alternatives. The final box represents the evaluation component of the P&G framework that involves the comparison of NED costs and benefits, and non-NED effects of project plans.

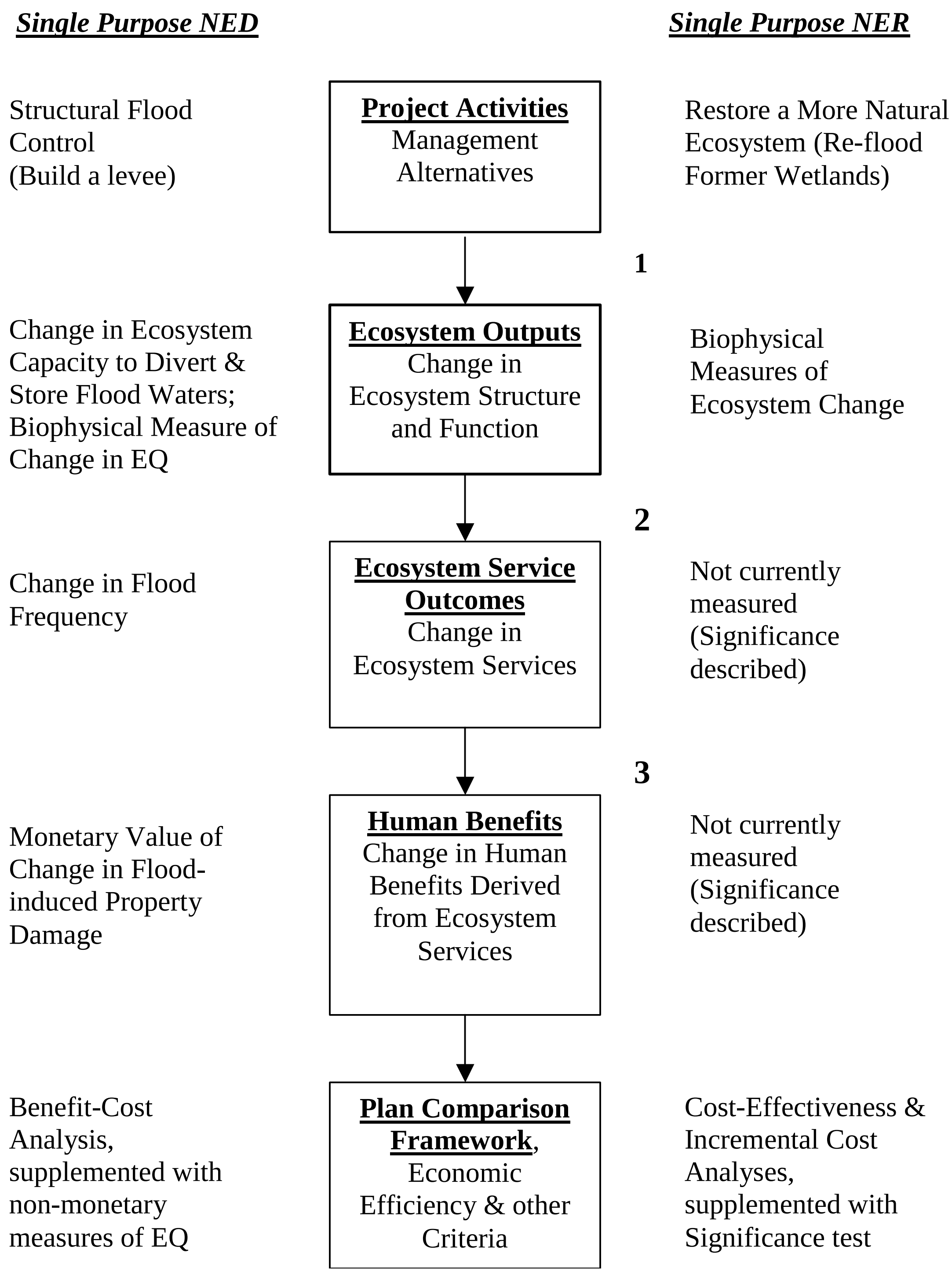



**Figure 5.2. Analysis and Evaluation of Single Purpose NED and NER Projects**

The NED project example involves structural flood control where the intended service outcome is urban flood hazard reduction. In this case the various linkages among project activities and NED benefits associated with the flood control service are all measurable. The first linkage establishes the increase in floodwater storage and diversion capacity expected to result from the flood control measures of alternative plans. This measure of ecosystem output provides the information needed to estimate the expected change in flood protection, the intended service outcome. The final linkage measures the economic value of this outcome based on the market value of flood damages avoided.

The flood control project also is shown to be associated with certain environmental effects, and for which the full set of linkages among management measures and NED (monetary) effects are not measured. These environmental effects are instead measured in terms of physical/biological metrics of expected changes and recorded in the EQ account.

The analytical results for all plans formulated in the flood control example provide the information needed to calculate and compare the estimated net NED benefits (monetary value of services yielded less project costs) of alternative plans. The estimated non-monetary EQ effects might also serve some limited role in the plan evaluation and selection. For example, measured EQ effects might be used to determine what mitigation measures are appropriate for each plan, the cost of which would be included in total plan costs. Moreover, a plan showing the highest net NED benefits (with mitigation costs taken into account) could be passed over for an alternative plan associated with less negative EQ effects. However, the recommended plan must be one for which estimated net NED benefits are positive.

In the NER project example, the planning objective relates directly to the types of environmental effects that play only a supplemental role (through the EQ account) in the evaluation and selection of the NED project. And while the flood control project focuses on one intended service outcome, the NER project might be pursued for a variety of related natural service outcomes. However, since it is not readily possible to estimate economic benefits for these services, a non-monetary measure of NER output based on predicted changes in ecosystem outputs is used as a proxy measure for NER benefits. That is, in this case the linkages among project activities, ecosystem outputs, service outcomes, and human benefits are not all estimated. Instead, the economic efficiency implications of and tradeoffs among alternative plans are determined by comparing plans in terms of their costs and non-monetary NER output using CE/IC analyses. Planners can then recommend a plan from among the cost effective set based on a subjective judgment that the level of non-monetary restoration outputs justify the cost to produce them. Corps guidance gives little insight into how that should be done, apart from specifying that restoration outputs must be shown to be “significant” based on institutional, public or technical recognition of importance.

### 5.4 Multipurpose NED/NER Projects

For multipurpose NED/NER projects, the PGN says that plan selection shall attempt to maximize the difference between the sum of NED and NER benefits and project costs, and to strike the best balance between the two objectives. As in the single purpose NER context, this justification standard necessarily requires a subjective determination of the "best" plan since NER outputs are measured in non-monetary terms.

The PGN suggests that the evaluation and comparison of NED/NER plans should rely on a combination of benefit-cost analysis and CE/IC analysis. Appendix E of the PGN explains that benefit-cost analysis should be used to relate NED benefits against that portion of plan costs required to produce these benefits, and CE/IC should be used to relate non-monetary NER outputs against that portion of plan costs required to produce those outputs. It also says that any joint costs, defined as plan costs that simultaneously produce both NED benefits and NER outputs, should be allocated among these purposes using the standard method used by the Corps for allocating costs to the various project sponsors for a plan selected for funding. (Cost allocation for purposes of cost sharing the selected plan is needed because Corps policy defines cost sharing rules that vary by project purpose.)

Joint costs should be the norm for multipurpose NED/NER projects since the primary rationale for pursing a multipurpose project instead of separate single purpose projects is efficiencies realized by exploiting opportunities to jointly produce desired outputs. For example, joint costs would make up the bulk of total costs for a project plan involving the use of floodplain evacuation to simultaneously serve flood control and NER purposes. In this case the costs of securing the required land and relocating structures people would serve both project objectives.

But the allocation of joint costs among project outputs *for the purpose of analyzing the economic efficiency implications of alternative plans* cannot be justified on economic grounds, and at any rate is not necessary nor helpful for that purpose. If a dollar's worth of plan cost serves both NED and NER outputs, these costs and benefits must be considered together for plan comparison.

This can be readily accomplished since plan costs and NED benefits are both measured in dollars and are recognized by Corps regulations as fungible (i.e., a dollar's worth of NED benefit for a formulated purpose exactly offsets a dollar's worth of plan implementation cost). Given this, the CE/IC framework is appropriate for analyzing the efficiency implications of multipurpose NED/NER project plans involving joint costs. In this case, the CE/IC procedures can be implemented using a measure of plan costs calculated by subtracting NED benefits yielded by some plan from the financial costs needed to implement that plan. That is, the two plan effects under consideration that are expressed in dollars would be combined into a "net cost" measure for each alternative plan. Net costs would then be used together with the non-monetary NER output measure to implement CE/IC analyses (see Figure 5.3).

Of course, implementing CE/IC analyses using a net cost measure masks information on the specific levels of implementation costs and NED benefits of plans. But its main purpose is efficiency analysis; that is, it serves to help weed out inefficient (dominated) plans from further consideration. The next step for tradeoff analysis would break out and compare all available information on project effects for the narrowed set of plans, as shown in Table 5.2.

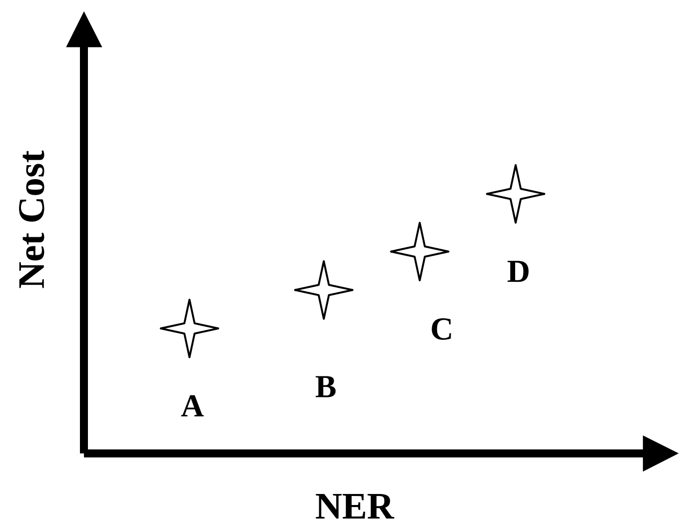



**Figure 5.3 Net Costs (implementation costs less NED benefits gained) of Alternative Plans Relative to NER Outputs.**

**Table 5.2 Display of Flood Damage Reduction (FDR) benefits, Recreation (Rec) benefits, Environmental (Env) benefits, and Implementation Costs for Cost-effective Plans**

| Plan Alternative | FDR Benefits | Rec Benefits | Costs | Env. Benefits |
|---|---|---|---|---|
| Plan A | $ | $ | $ | Non-monetary Output Measure |
| Plan B | $ | $ | $ | Non-monetary Output Measure |
| Plan C | $ | $ | $ | Non-monetary Output Measure |
| Plan D | $ | $ | $ | Non-monetary Output Measure |

Current Corps policy guidance does not speak to the use of cost-effectiveness analysis for examining tradeoffs between the net economic development and environmental effects of alternative plans. But it is worth noting that this same basic framework was once used for a short time in the Corps history. Figure 5.4 shows an example of a formal NED-EQ tradeoff analysis developed for a navigation project under consideration in 1977. At that time the Corps planning rules in effect, the *Principles and Standards*, required the formulation of plans that maximized net NED benefits (the NED plan), as well as plans that maximized environmental quality (the EQ plan), however it was defined and measured at that time.

The tradeoff graph shown in Figure 5.4 is equivalent to the cost effectiveness graph discussed above although it differs in perspective. In the NED-NER tradeoff graph, the vertical axis from the origin upward shows positive net NED (i.e., net dollar benefits), while in the CE graph this portion of the vertical axis shows negative net NED (i.e., net costs). Despite this different perspective, both graphs compare the same thing--NER output with *net* NED effects.

Figure 5.5 presents another example of a tradeoff analysis between net NED benefits and some measure of ecological quality developed for an actual project study. The project in this case examined the implications of restoring natural flow variability of a river system, where environmental effects were measured using an ecological index devised by the Nature Conservancy. Different combinations of reservoir operating rules for the managed system were developed, each addressing a different set of water management objectives (e.g. maximize recreation, navigation or environmental quality). In this project example, the best environmental result achievable was determined to be something far less than the "ideal natural state" because of other man-made alterations to the river system. Further, the tradeoff analysis showed that, in order to achieve this level of ecological quality, nearly all the economic benefits for other multiple purposes (navigation, hydropower, water supply, recreation, etc.) would have to be foregone. Hence, this analysis served to illustrate the opportunity costs in terms of lost economic development opportunities associated with the choice of reservoir operating rules designed to serve environmental quality objectives.

A final point on the use of CE/IC (or NED-NER tradeoff) analysis for multipurpose planning should be recognized. Some within the Corps have expressed concern that the subjective justification standard applicable to NED/NER planning could be abused. The specific concern noted is that NED-focused plans which would otherwise fail the benefit-cost test required for a single purpose NED project, and that do not also produce a significant level of NER output (i.e., that involve little joint production), could be combined with largely separable NER features and show up on the cost-effectiveness frontier in a multipurpose planning case. Planners would then have the opportunity to select these plans following the subjective justification standard applicable to NED/NER planning. This could provide an avenue to push forward NED-focused plans that could not be justified on their own, by simply adding on some NER-focused features.

This is legitimate concern, although one that has long been recognized and addressed by Corps planning rules for traditional (NED only) multipurpose planning. In that context, each purpose represented in a justified plan (i.e., one for which total NED benefits exceed total costs) must be incrementally justified. Incremental justification requires that purpose-specific dollar benefits, as limited by the cost of the least-cost alternative single purpose plan providing equivalent benefits, must equal or exceed separable costs for that purpose, where separable costs are defined as the cost of the multipurpose plan with that purpose included less cost of the plan with that purpose omitted. The incremental justification test ensures that each purpose in a NED-only multipurpose plan adds to rather that subtracts from total net benefits produced by that plan.

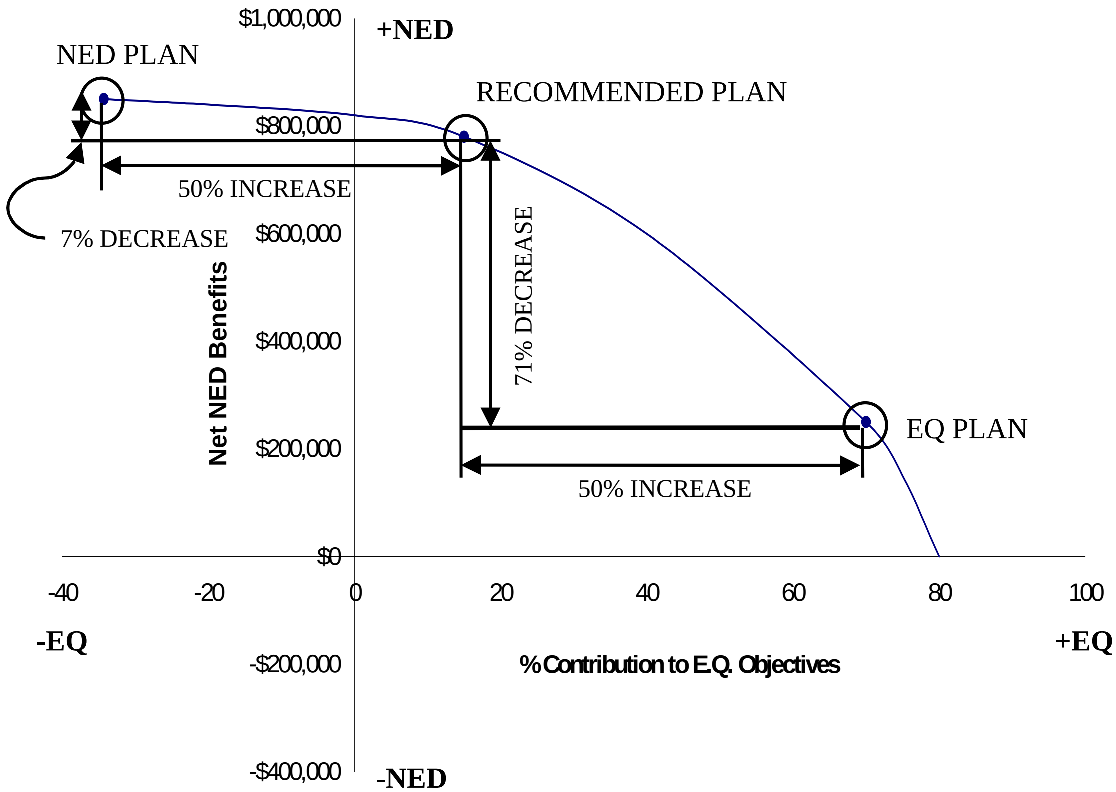
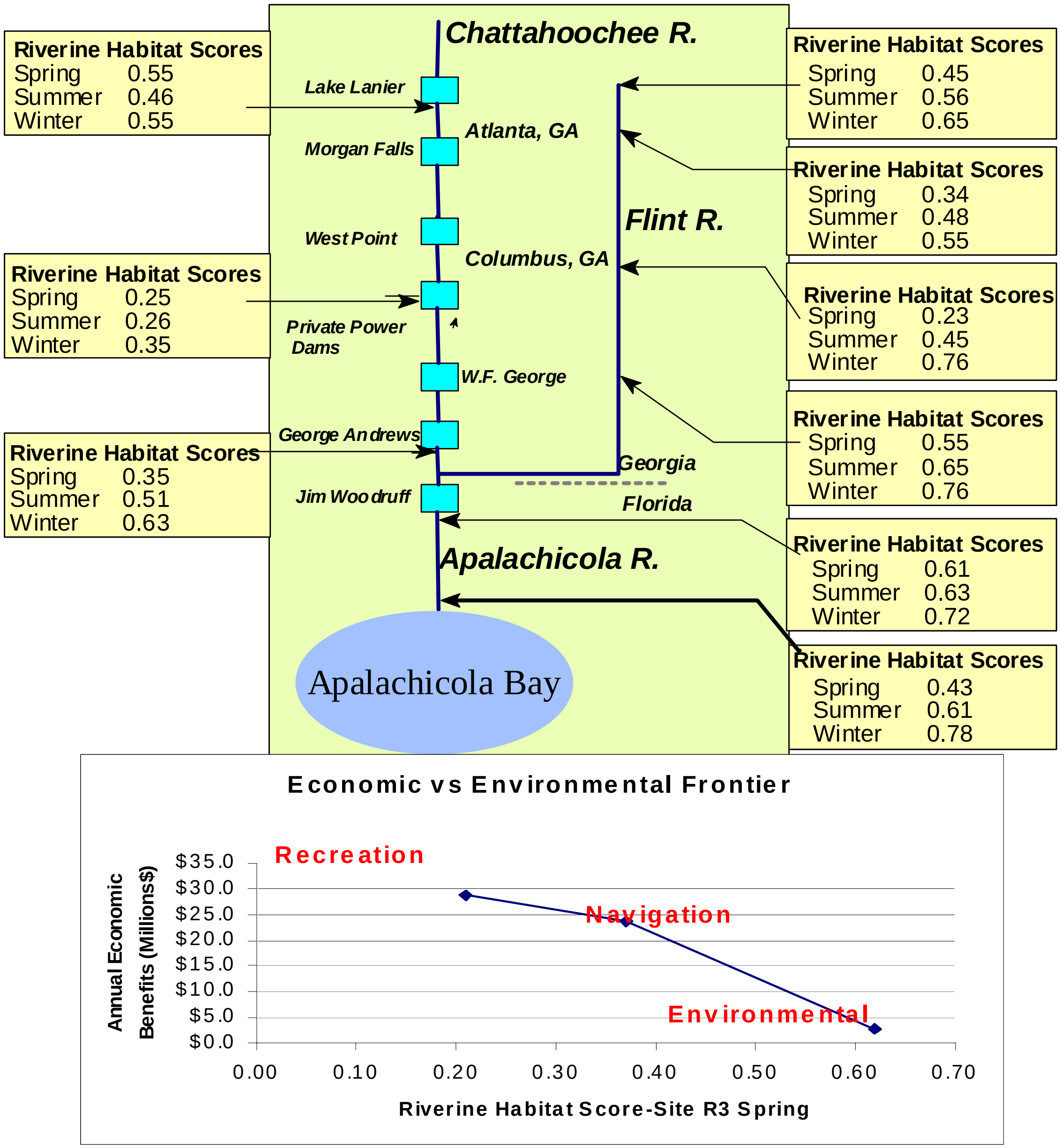



**Figure 5.4 NED-EQ Tradeoff for Wilmington Navigation Project**

## Figure 5.5 Consideration of EQ-NED Trade-Offs




The Schematic diagram of ACF reservoirs and river reaches (left) shows measuring points for three types of environmental impacts, Riverine Habitat, Riparian Wetlands and Reservoir Fisheries. A consensus of study biologists agreed that the best single metric for ranking the environmental desirability of flow regimes was the Riverine habitat measure at site R3.

The graph (lower left) shows a tradeoff between economic benefits (y-axis) and environmental desirability (x-axis) for three alternative reservoir-operating plans. The **Recreation** alternative minimizes releases to keep reservoirs high for boating. The **Environmental** alternative eliminates reservoir regulation. The **Navigation** alternative draws reservoirs down during drought to maintain downstream flows and navigation depths. The graph shows that the Navigation alternative splits the difference in the environmental score at a cost of about $5 million per year. Choosing “Environmental” over “Navigation” provides the same incremental environmental gain at a cost of $23 million.

In the NED/NER case, the incremental justification test would show that NED/NER plans formed by combining NER-specific plan features with NED features that could not otherwise be justified as single purpose NED plans and that do not also provide significant NER output, are not justifiable as multipurpose plans even if they show up on the CE frontier (i.e., represent a non-dominated plan). In this context the incremental justification test ensures that multipurpose plans are providing efficiencies over two or more single-purpose plans serving each purpose (output) individually. The key to achieving such efficiencies is the degree to which NED/NER plans involve joint production, as indicated by the extent to which plan costs jointly produce both NED and NER output. The greater joint costs are in relation to total plan costs, the easier it would be for each purpose in a NED/NER plan to be incrementally justified.

In sum, the incremental justification test, if correctly applied, should prevent abuse of the subjective justification standard applicable to multipurpose planning by exposing plans that do not involve more than a trivial level of joint production. At the same time, the subjective justification standard applicable to NED/NER planning offers the opportunity to justify multipurpose plans that involve significant joint production. Thus, for example, a floodplain evacuation plan that otherwise could not be justified as a single purpose flood control project, could be justified as a multipurpose NED/NER plan to the extent that it jointly produces a significant level of NER output.

### 5.5 Foregone and Incidental Benefits

Civil Works plans sometimes involve foregone and/or incidental benefits that are unrelated to project objectives, and that can be valued in monetary terms. Foregone benefits are the opportunity costs associated with a reduction of current levels of NED services expected to result from project plans. Incidental benefits are the value of expected NED outputs that are different from the specific outputs for which plans are formulated, and for which no additional project expenditure is required. In the case of a single-purpose flood control project, for example, any existing recreation benefits lost due to project plans would be viewed as foregone benefits, while any added recreational benefits yielded would be viewed as incidental benefits. Although they represent two sides of the same coin, Corps rules treat foregone benefits differently from incidental benefits for project evaluation, comparison and justification (see Table 5.2).

Corps regulations say that the estimation of plan costs should include any foregone NED benefits of plans. These opportunity costs thus would be appropriately included in the cost measure used for CE/IC analysis. For example, if a single-purpose NER project plan resulted in a reduction in an existing flood control service, then these lost NED benefits would be estimated and added to plan implementation costs to calculate total plan costs. Foregone benefits thus would be considered directly within the CE/IC framework used to evaluate the economic efficiency implications of alternative restoration plans.

**Table 5.2 Corps Planning Guidance on Foregone & Incidental Benefits ***

| | |
|---|---|
| **Project Costs** | Project measures, whether structural or nonstructural, require the use of various resources. NED costs are used for the economic analysis of alternative projects and reflect the opportunity costs of direct or indirect resources consumed by project implementation. From an economic perspective, the real measure of cost is opportunity cost, i.e., the value of that which is foregone when a choice of a particular plan is made. In order to capture the opportunity costs of proposed plans, NED costs include three types of costs: implementation costs, other direct costs and associated costs." … "Other direct costs are the costs of resources directly required for a project or a plan but for which no implementation outlays are made. Examples of these costs are interest during construction, value of donated land, *uncompensated NED losses and other negative externalities*." [Italics added] Source: PGN Section 2-2k |
| **Project Benefits** | "Ecosystem restoration outputs must be clearly identified and quantified in appropriate units. Although it is possible to evaluate various physical, chemical and/or biological parameters that can be modified by management measures which would result in an increase in ecosystem quantity and quality in the project area, the use of units that measure an increase in "ecosystem" value and productivity are preferred"… "*Monetary gains (e.g., incidental recreation or flood damage reduction) and losses (e.g., flood damage reduction or hydropower) associated with the project shall also be identified*." [Italics added] Source: PGN Section 3-5c(1) |
| **Evaluation Focus** | "While the planning process for single purpose ecosystem restoration projects is the same as for any other purpose, the evaluation process is somewhat different in that it focuses on quantitative and qualitative restoration outputs and *monetary benefits are usually incidental*." [Italics added] Source: PGN Section 3-5c |

***** Source: *Planning Guidance Notebook* (ER 1105-2-100; April 22, 2000).

At the same time, Corps rules suggest that the incidental NED benefits of restoration project plans should *not* be combined with plan implementation costs for plan evaluation within the CE/IC framework. That is, Corps policy seems to disallow plan comparison and justification based on CE/IC evaluations that use a net measure of plan costs calculated by subtracting the incidental benefits of plans from plan implementation costs. The reasoning is that such a net cost measure could obscure information needed to ensure that the Federal interest in priority outputs are served by recommended plans. For example, a local sponsor intent on gaining approval for a NER project pursued by the locality primarily for recreation services might want to define and use a measure of project costs net of estimated monetary recreation benefits yielded to help show the project is justified. To avoid this possibility, Corps policy suggests that recommended plans must be shown to be cost-effective based solely on the comparison of plan costs (including foregone NED benefits) and the non-monetary measure of NER output. Any estimated incidental NED benefits could serve a supplemental role in the determination of project worth, but not a direct role in the CE/IC analyses used for plan comparison and justification.

This procedure is consistent with the way that incidental benefits are treated in the evaluation of NED projects, at least in the case of some authorized Civil Works purposes.

For example, a recommended structural flood control project plan must be shown to produce flood hazard reduction benefits in excess of project costs; any estimated incidental recreation benefits associated with the project could not be used to meet this positive net benefits standard for project justification.

In the restoration context, data on monetary benefits can usefully inform plan selection without being included within CE/IC analyses. Specifically, this data can serve as supplemental information when using incremental costs analysis to help decide the "is it worth it" question for cost-effective plans that provide successively higher levels of NER output (Shabman, 1993).

### 5.6 Discounting and Plan Evaluation

The costs and benefits of Civil Works projects are typically paid and received at different times throughout the project time horizon. For example, construction costs are incurred in the initial years of a project, while operation and maintenance costs are paid and project benefits are realized as annual flows throughout the project useful life. In order to inform present-day public investment decisions, project evaluation requires that project costs and benefits be translated into comparable present-day values.

"Discounting" is the method by which project costs and benefits that occur in different time periods are adjusted to reflect that a given amount of consumption in some future time period is worth less than the same amount of consumption today. Essentially, discounting is an added valuation process that measures the "time value" of project costs and benefits.

Discounting project costs and benefits that are expressed in dollar terms is relatively straightforward and uncontroversial with respect to the evaluation of public investments that affect only present-day generations (although choice of the appropriate interest rate for discounting project effects over time remains highly contentious). The same is not true with respect to project effects that are estimated in non-monetary terms, however, such as will be the case for ecosystem restoration outputs. There is generally no consensus on whether it is appropriate to discount non-monetary effects of public investment decisions for project evaluation.

One view holds that project effects that are measured in non-monetary terms and that do not have a close connection to service outcomes and monetary benefits should not be discounted for project evaluation. For example, the measurement of ecosystem restoration outputs generally must rely on some measure of ecosystem function as a gross proxy for "natural" ecosystem service outcomes. But since this functional measure does not directly say anything about the magnitude or timing of natural service flows or associated benefits, it should not be discounted for project evaluation.

The PGN seems to adopt this view by specifying that non-monetary ecosystem restoration outputs should not be discounted for project evaluation. Instead, it says that these output measures should be computed as average annual measures, taking into consideration that the outputs of alternative plans are likely to vary over time. For example, consider two restoration plans that each produces 50 functional units annually when restoration outputs are fully realized. If the first plan achieves the full 50 functional units in year 1 after project construction, while the other will take 10 years of gradually increasing output to reach the 50 functional units, then this information should inform the calculation of average annual output for the two plans. In this example the first plan would produce an average annual output of 50 functional units over the project life, while the second would produce something less. This highlights that information on the timing of non-monetary outputs is always relevant for project evaluation and thus should be considered in some way.

## Section 6. Possibilities for Monetary Evaluation of Restoration Outputs

In Section 5 it was asserted that the concept of economic (or NED) value is applicable to the broad suite of services affected by Civil Works projects, including those that are most closely aligned with natural ecosystem parts and processes. This suggests that, to the extent that acceptable monetary estimates of restoration outputs could be practically generated for project evaluation, the monetary standard used for evaluating and justifying traditional Civil Works projects could also be applied to ecosystem restoration projects (National Research Council, 1999a). This section briefly explores technical and conceptual issues relating to the possibility for using a monetary evaluation standard for restoration project planning.

### 6.1 Definition of Economic Value

The concept of economic value, as defined by neoclassical welfare economics, follows from the premise that each person is the relevant judge of what is "good" for that individual based on the degree to which his or her preferences are satisfied. The theory assumes that each person has well-defined and stable preferences for alternative bundles of goods and services that include goods that are exchanged in the marketplace (market goods) and goods that are not (non-market goods). And importantly, it is assumed that there is broad scope for substitution among goods in the pursuit of preference satisfaction. This implies that the effect of a decrease (increase) in the consumption of some good on an individual's level of preference satisfaction can be offset through an increase (decrease) in the consumption of other goods (Freeman, 1993).

The concept of economic value rests squarely on this "utilitarian" premise that human welfare derives from preference satisfaction. Acceptance of that premise implies that the tradeoffs that a person makes as he or she chooses less of one good in favor of more of another good reveals something about the value of this tradeoff to the individual. Formally, the economic value of some change (tradeoff) to an affected individual is defined as the amount of monetary compensation (positive or negative) that the individual would need in order to maintain the same level of individual preference satisfaction with the change as without the change. This measure of compensation is specific to each affected individual and is entirely dependent on the circumstances of the specific change context (Bockstael, et al., 1998).

For example, consider a policy proposal to newly allow hunting in some public wildlife area. An affected individual who is a hunting enthusiast might be expected to realize an increase in preference satisfaction if the policy were implemented (although this result would depend on the supply and quality of other hunting sites in the same general vicinity as well as other circumstances specific to the change context and the individual). If the policy were implemented to this person's benefit, he or she would require negative compensation, as represented by the individual's maximum willingness to pay for the opportunity to hunt in the wildlife area, in order to maintain the same level of individual welfare experienced in the absence of that opportunity. This "willingness to pay" (WTP) measure of compensation reflects the measurement standard defined by the P&G for evaluating the NED benefits of water resource projects; total project benefits are defined as the sum of WTP for each individual who stands to gain from the project.

Now consider another affected individual who is not a hunter but who enjoys bird watching in the wildlife area being considered for hunting use. This person might be expected to experience a decreased level of preference satisfaction if the area were opened to hunting. In that event, this person would require positive compensation, as represented by the minimum amount of money the individual would willingly accept to bear costs resulting from the hunting policy, in order to maintain the same level of individual welfare with the policy in place as without the policy. This “willingness-to-accept” (WTA) measure of compensation reflects the measurement standard defined by the P&G for the evaluation of the NED costs of water resource projects; total project costs are defined as the sum of WTA for each individual who stands to lose from the project.

### 6.2 Measurement of Economic Value

The marketplace provides the context for inferring economic values since the market price for some good provides a dollar measure of the amount of other goods that would need to be reduced in order to purchase it. Thus, for a marketed good, observed variations between market price and quantity consumed provide the basis for estimating the demand function (marginal WTP function) for that good. This demand function provides the information needed to estimate the economic value of structural changes in the supply and/or quality of the good.

For a variety of reasons, most ecosystem services are not traded in competitive markets, so there are no associated price data providing a basis for valuation. To address this problem, economic methods have been developed to estimate “shadow prices” for non-market goods that, in theory, represent the market prices that would emerge if these goods were traded in competitive markets. One class of methods, referred to as “revealed preferences” approaches, attempt to reveal shadow prices by examining market data on marketed goods that are linked in some way to the non-market good. Another class of valuation methods, referred to as “stated preferences” approaches, have been developed and applied to situations in which the market choices of people provide insufficient clues about their preferences for non-market goods.

### 6.3 Monetary Evaluation of Traditional Outputs

The Corps has long faced the need to use non-market valuation tools since traditional Civil Works outputs generally are not traded in competitive markets (Table 6.1 provides an overview of valuation techniques specified in the P&G). However, most traditional outputs have close market counterparts that facilitate valuation based on *change in net income* or *cost of most likely alternative*. So, for example, the benefits from enhancing waterway transportation links are assessed in terms of costs savings to commercial navigation shippers, and the benefits from enhancing flood regulation services are assessed in terms of property damages avoided. Similarly, the benefits of introducing new sources of water supply and hydropower are estimated based on the cost of providing equivalent outputs using the least-cost alternative source. This valuation approach follows from the recognition that the affected population would be forced to obtain alternative sources of these outputs if the project source was not forthcoming.

In general, the valuation of traditional Civil Works outputs such as commercial navigation, flood damage reduction, hydropower and water supply has been readily possible for two main reasons. First, project-induced alterations in the underlying ecosystem service flows (e.g., waterway transportation capacity, flood storage and diversion capacity) are intensive and largely involve physical relationships that are well understood and predictable. Thus, for these traditional outputs, the types of non-economic information on service flows needed for valuation is readily obtained. Second, as outlined above, these outputs generally have close market counterparts that provide market evidence for benefits assessment.

**Table 6.1. Broad Approaches and Specific Techniques for Economic Valuation Specified by the Principles & Guidelines (P&G)**

| **P&G Approaches for NED Estimation** | **Specific Non-Market Valuation Techniques** | **Applicable Benefit Categories** |
|---|---|---|
| Change in Net Income | • Factor Income/Avoided Costs<br>• Property Damages Avoided | Market productivity of ecological systems in production/consumption (e.g., inland navigation, flood hazard reduction) |
| Cost of Most Likely Alternative | • Replacement Cost | Service replacement (e.g., electricity, water supply) |
| Simulated Market Price | • Travel Cost<br>• Hedonic Property Value<br>• Contingent Valuation | Utility derived from direct use of ecological amenities (e.g., recreation) |
| Administratively Established Values | • Unit Day Values | Utility derived from certain recreational uses (e.g., hunting and fishing) |

Recreation, on the other hand, represents a traditional Civil Works output that generally has no close market counterpart providing direct evidence for benefits assessment. Corps guidance sets out a set of techniques for estimating *simulated market price* (i.e., shadow prices) as the basis for assessing recreation benefits. These include, for example, the Travel Cost methods that looks to indirect evidence of shadow prices based on the time and money people spend to visit a recreation site. Perhaps in recognition that these methods can be difficult and costly to implement, however, Corps guidance also allows project recreation benefits to evaluated using *administratively established values* that represent average unit values for a day of fishing or hunting derived from previous studies.

## 6.4 Monetary Evaluation of Restoration Outputs

### 6.4.1 Technical Issues

In general, the specific techniques recommended by the P&G for valuing traditional outputs are also applicable to the types of "natural" ecosystem service outputs likely to be associated with ecosystem restoration. (Table 6.2 summarizes the general applications, evaluation basis, and strength and limits of these techniques in the restoration context.) This does not mean that valuation prospects for project-induced changes in natural services are generally favorable, however. One reason is that the non-economic relationships between management action and natural service outputs often represent complex hydrological and biological relationships that are not well understood and readily predictable (especially in situations requiring long periods of restoration time in environments susceptible to many future uncontrollable human impacts). A second reason is that natural services often directly affect the quality of human life in ways that have no close connection to the use of market goods. As discussed below for different types of natural service benefits, these factors pose significant technical limitations for the economic valuation of restoration outputs.

Economic valuation based on replacement cost may be appropriate when restoration efforts affect traditional outputs such as water supply and hydropower. However, attempts to estimate values for changes in natural ecosystem service outputs (e.g., waste treatment) based on the cost of replacing the service with a human-engineered alternative often founder because they fail to provide evidence that the alternative cost would actually be incurred if the natural service were not available. [See, for example, attempts by Costanza, et al. (1997) to use replacement cost as a measure of benefit for oceanic nutrient storage].

Ecosystem restoration might sometimes positively or negatively affect ecosystem services that serve as inputs into the production of marketed goods. When these effects involve traditional Civil Works outputs such as agricultural water supply, for example, they generally could be quantified and valued using P&G methods. However, valuation is much more difficult and limited by current knowledge and data when service outputs are farther removed from the end product of market value. Consider, for example, the contribution of estuarine wetlands to marine fisheries as a provider of food and nursery habitat. In this case the valuation of changes in the habitat service requires tracing through complex and uncertain bio-economic relationships among management action, wetland habitat, fish stocks, and fishery productivity.

Restoration might be expected to often affect recreation uses of ecosystems, for which various revealed preference techniques have been specifically developed and refined over the last several decades. Recreation benefits are the one class of restoration outputs that might be most readily valued in dollar terms, although even this case poses significant technical challenges for specifying and estimating the linkages among restoration actions and recreation behavior. Moreover, region-wide modeling would generally be needed

**Table 6.2 Overview of Non-market Techniques for Valuing Restoration Outputs**

| Technique | General Applications | Measurement Basis | Major Strengths and Limits |
|---|---|---|---|
| Factor Income/ Avoided Costs | Use values for ecosystem services that serve as factors of production for market goods | Relies on estimating and using production relationships for the marketed good to infer how changes in ecosystem services will affect the profits or costs of producers | Main strength is that it avoids the need to estimate demand for the market good. However, the supply side focus is reasonable only if the production unit in question is small relative to the overall production of the market good, or if the improvement is ecosystem service input represents only a marginal change. |
| Property Damages Avoided | Use values for flood risk reduction & other ecosystem services that prevent property damage | Relies on estimating repair costs to specific properties with and without flood risk reduction services | Main strength is that value estimates are relatively easily, consistently and inexpensively made. Main limitation is that value estimates are hypothetical, since no post-damage repair choices are observed. Also, value estimates reflect only one potential dimension of willingness to pay. |
| Travel Cost | Use values for recreational uses of ecosystems | Investigates changes in the quantities consumed of a complementary market good, travel to the site, to estimate demand for site recreational uses | Main strength is that value estimates are based on the actual choices of people. One limitation is that region-wide modeling would generally be needed to estimate the implications for benefits of changes in site quality |
| Hedonic Property Value | Use values for location-specific ecosystem amenities and services that prevent property damage | Investigates prices of a complementary market good, residential property, to reveal implicit prices for location-specific ecosystem amenities or damage prevention services | Main strength is that value estimates are based on the actual choices of people. One limitation is that the scope of ecosystem values that can be estimated is limited to the set of ecosystem services that can be captured by people through their choice of residential location |
| Contingent Valuation | Use and passive use values for ecosystem services that affect human welfare in ways other than through market production | Relies on the use of sophisticated surveys to elicit information from respondents on their preferences for ecosystem services | Main strength is its flexibility that allows it to be used to estimate passive use benefits as well as use benefits associated with ecosystem services individually and in combination. Main limitation is that responses to hypothetical questions may not reflect what people would actually pay for ecosystem services in a real economic or policy choice setting. |
| Benefits Transfer | Use values for recreational uses of ecosystems | Relies on valuation results for some site(s) derived in previous studies (e.g., unit day values) to develop value estimates for the project site | Main strength is that it can be applied quickly and inexpensively. Main limitation is that it can provide only a gross approximation of benefits at project sites since recreational values are context (e.g., site, user) specific. Also not well suited to assessing benefits from changes in site quality. |
| Replacement Cost | Use values for ecosystem services that can be provided through alternative means | Relies on estimates of the cost of most economical alternative means for providing equivalent services | Its main strength, that it avoids estimation of the links between ecosystem services and human welfare, is also its major limitation. Can approximate service value only if 1) the replacement provides the same function at the same level as the ecosystem service, and 2) evidence suggests that people would be willing to incur this cost if the service were not available. |

to estimate site-specific recreation benefits in order to account for available substitute sites.

Valuation prospects are much more limited for changes in natural ecosystem services that may contribute to human welfare independent of human use. Service outputs relating to the restoration of natural biological diversity for the sustenance of endangered species, for example, might give rise to such "passive use" benefits (as well as possible use values). The only available valuation techniques for estimating passive use benefits are stated preference methods. For example, the most common such technique, the Contingent Valuation Method (CVM), relies on the use of sophisticated surveys to get individuals to express their preferences for non-market services through money bids in simulated markets, policy referenda, or other hypothetical choice settings. Typically, a referendum format is used to elicit preferences for environmental protection or restoration programs. For example, respondents are asked if they would vote for an environmental management regime at a cost of $X to the respondent. In the survey, the amount of $X varies across respondents, enabling researchers to trace out a demand function from which willingness to pay can be derived.

The great advantage of CVM is its flexibility that facilitates its use to elicit use and passive use values associated with the improvement of many types of ecosystem services, individually and collectively. However, such valuation depends on the ability to forecast how projects might affect ecosystem attributes and convey this information to survey respondents in terms that are meaningful to them. Moreover, use of CVM for estimating environmental benefits remains controversial and not universally accepted within the economics profession since it produces value estimates that are not based on the actual choices of people. Its use for estimating passive use values is particularly controversial since there is no way to verify valuation results. Further, the number of people that may hold passive use values for natural resources with public goods characteristics is not known, and relatively small estimated values for a representative individual, when applied to large populations, can result in very high estimates of resource value. Such high valuation results feed the skepticism of those in the economics community and others who question the adequacy of hypothetical choice methods for valuing ecosystem services.

The above review suggests that, in general, the monetary valuation of non-market ecosystem service outcomes that are far removed from the end product of market value, or that directly affect the quality of human life, is severely limited by technical hurdles. Professor A. Myrick Freeman, in the concluding chapter to his 1993 book on the state-of-the art in measuring environmental and resource values, writes:

> The economic framework, with its focus on the welfare of humans, is inadequate to the task of valuing such things as biodiversity, the reduction of ecological risks, and the protection of basic ecosystem functions. When policies to protect biodiversity or ecosystems are proposed, economists may be able to say something sensible about the costs of those policies, but except where nonuse

> values are involved or where people use ecosystems (for example, for commercial harvesting of fish or for recreation), economists will not be able to contribute comparable welfare measures on the benefit side of the equation. (Freeman, 1993, p.485)

Professor Freeman's pessimism regarding prospects for valuing changes in certain types of ecosystem services may spring at least in part from doubt on his part that the obstacles to establishing the non-economic foundations of valuation can be overcome. After all, neither the actions of individuals in the marketplace or their responses to WTP surveys can reveal meaningful values for changes in ecosystem services if these individuals do not understand how these services contribute to preference satisfaction (Bockstael, et al, 1998).

But Freeman's remarks also seem to cast doubt on the validity of the economic concept of value as it relates to certain types of natural ecosystem services--indeed, the very types that may often be the focus of efforts to restore natural ecosystem parts and processes. Conceptual controversies surrounding the economic basis for measuring and using ecosystem service values for guiding public decision making are outlined briefly below.

### 6.4.2 Conceptual Issues

Critics of using valuation to guide environmental policy making can be found within the economics profession as well as among philosophers, psychologists and political scientists. These critics question whether the choices that people make in markets or hypothetical choice contexts can be interpreted as a reflection of well-defined and stable human preferences, or whether any such interpretations provide an appropriate basis for guiding environmental investments or regulations.

Professor Leonard Shabman and colleagues have summarized controversies surrounding these propositions from within the economics profession (Shabman and Stephenson, 2000; Shabman, 1993). They outline two main strains of economic thought challenge the notion that the economic concept of value is relevant or appropriate for guiding environmental decision making.

One comes from the Austrian school of economic thought that advances an interpretation of the role of market exchange as one of preference discovery and revision. According to the Austrian economists, the market choices of people are not dictated by a set of fixed preferences that are exogenously determined (i.e., determined independently of the choice context). Rather, the Austrian view is that an individual's preferences are endogenously determined by his or her knowledge of available choices at any given time, and these preferences are subject to continuous change as the individual gains more information about and experience with goods and their alternatives, and as personal circumstances change. Acceptance of the Austrian view that market exchange is a process by which individuals continually discover and revise preferences implies that market prices cannot be used as datum to reveal meaningful values for ecosystem services.

Research by behavioral psychologists on how people make decisions lends support to the view that people do not retrieve previously determined preferences when making complex choices. Psychologists have voiced the view that when people are faced with choices made under unfamiliar conditions and with limited information, the choices observed are not dictated by retrieved preferences, but rather preferences that are constructed at the time based on the choice context and circumstances (Schkade, 1994). This is a particularly important criticism for the use of CVM questions to elicit values for ecosystem services, one that has been made by psychologists as well as some economists.

The other major economic criticism for valuing ecosystem services as a guide to environmental policy comes from the so-called Institutional economists. The main focus of the Institutional economists critique is on the use of the positive net benefits criterion (i.e., national economic efficiency standard), based on the summation of individuals' economic gains and losses, for guiding policy making. Institutional economists note that observed market choices and prices reflect the distribution of income as much as individual preferences, and thus raise distributional concerns. In the words of Shabman and Stephenson (2000), "the institutional economists argue that non-market valuation inappropriately elevates the preferences of current individuals and those with the greatest income (ability to pay) to the touchstone of environmental decision making."

More generally, use of the efficiency standard for justifying public investments and regulations has long been a point of controversy within neoclassical welfare economics, although these concerns are now rarely voiced (Bockstael, et al, 1991; Just, et al. 1982). The efficiency standard is based on the premise that a public investment is in the national interest if those individuals who gain from the investment could fully compensate those individuals who lose, and still be better off. But since the efficiency of some investment is determined using benefit and cost measures that are conditioned upon the initial distribution of wealth, use of the efficiency standard for policy making implicitly assumes that the existing wealth distribution is desirable. This, of course, is debatable. That assumption, coupled with the fact that compensation is rarely paid to those who individuals who experience a loss from a public investment or regulation, raises serious concerns about the distributional effects over time of public decisions guided by the efficiency standard.

For the Institutional economist, such distributional concerns are particularly important in the case of environmental policy making since people often attach moral and social importance to environmental issues that they normally express through the political process, not through market choices. Given this, institutional economists argue that it is inappropriate to base environmental investment and regulatory decisions on preferences revealed from market exchange (Bromley, 1997).

Political scientists and philosophers have offered similar criticisms of the use of market prices for revealing human preferences for environmental and other investments that may involve a moral or community dimension. For example, Professor Arthur Maass in a

1966 paper on the relevance of benefit-cost analysis for guiding public investments decisions writes:

> "The second basic assumption of the new welfare economics and of benefit-cost analysis that needs to be challenged is consumers' sovereignty—reliance solely on market-exhibited preferences of individuals. This assumption…is not relevant to all public investment decisions, for an individual's market preference is a response in terms of what he believes to be good for his own economic interest, not for the community. Each individual plays a number of roles in his life…and each role can lead him to a unique response to a given situation. Thus, an individual has the capacity to respond to a given case, to formulate his preferences, in several ways, including these two: (1) what he believes to be good for himself—largely his economic self-interest, and (2) what he believes to be good for the political community. The difference between these two can be defined in terms of breadth of view. To the extent that an individual's response is community, rather than privately oriented, it places greater emphasis on the individual's estimate of the consequences of his choice on the larger community." (Maass, 1966)

Mark Sagoff, a professor of Philosophy, has advanced essentially the same argument about the different types of preferences that people hold, and he goes further to make judgments about the relevance of each for environmental policymaking. Professor Sagoff argues that people simultaneously hold "ideal-regarding preferences" that reflect community concerns and "self-regarding preferences" that reflect individual desires. In his view, the WTP concept of value is not relevant or appropriate for environmental policymaking since it is individuals' community-oriented preferences, not personal desires, that dominate the way in which people view environmental issues and judge protection policies (Sagoff, 1988).

### 6.5 Concluding Remarks on Monetary Evaluation of Restoration Outputs

The above review suggests that considerable technical obstacles, both non-economic and economic, stand in the way of comprehensive monetary accounting of restoration project benefits. Scientific obstacles relate to problems in tracing the links between restoration actions and service outcomes underlying all possible routes to human benefits. Economic obstacles relate to methodological limitations for measuring non-market benefits of service outcomes that affect the quality of human life in ways that have no close connection to marketed goods. Together, these obstacles to comprehensive valuation of restoration outputs impede use of a monetary standard for evaluating and justifying restoration projects.

In addition, some economists and other professionals have questioned the relevance of the economic concept of value as it relates to certain types of ecosystem services that might often be the focus of restoration projects. Challenges from these critics could hinder the political acceptability of using a monetary standard for evaluating and justifying restoration project plans, even if the technical hurdles to ecosystem valuation are significantly lowered over time.

Nevertheless, in some cases it should be technically possible to estimate monetary values for restoration outputs that could be used to inform project decisions in ways that are politically acceptable. For example, when restoration plans affect traditional Civil Works outputs such as hydroelectric generation and recreation, these effects could and should be estimated. If project plans involve a reduction in existing levels of traditional outputs, these opportunity costs could be included directly in the cost measure used to evaluate and compare plan alternatives against non-monetary measures of restoration outputs within a cost-effectiveness framework (Moser, 1990; Shabman, 1993). Similarly, in the multipurpose NED/NER context, estimated benefits for traditional outputs for which plans are formulated could be estimated and netted from the measure of plan implementation costs used for cost-effectiveness and incremental cost analyses.

When restoration project plans affect traditional or other service benefits that are unrelated to the specific outputs for which plans are formulated and that can be readily assessed in dollar terms, these “incidental benefits” should be estimated even though Corps policy may prevent their use for plan comparison directly within the CE/ICA framework. But such estimates could still serve a useful function as a sidebar to incremental cost analysis by helping to answer the “is it worth it” question for the set of plans identified as non-dominated based on the comparison of non-monetary measures of ecosystem outputs and monetary opportunity costs. In essence, such value estimates would provide one direct indication of the “significance” of restoration outputs.

## Section 7. Conclusions and Strategy for Improving Environmental Benefits Analysis

### 7.1 Conclusions.

During this study numerous issues associated with improving environmental benefits analysis for application in Civil Works studies were identified and examined. Among the conclusions that can be drawn from this report is that there is no "universal unit" for expressing ecosystem restoration benefits that is widely applicable across the full range of effects of restoration plans.

The study revealed numerous interrelated issues of ecology, economics, and evaluation that challenge the selection and development of environmental models, as well as improvements in environmental benefits analysis more generally. The s*cience relating system response to restoration measures* is better developed in principle than in specific applications. The *incorporation of ecological concepts into Corps policy, guidance and practice* is still evolving, and is becoming more complex as the Corps moves toward formulation of projects with combined economic and ecological outputs. For various reasons, Corps *planners have generally relied on a subset of available environmental assessment models* – mostly species-habitat index models – apparently because of inadequate scientific understanding and databases, past computing limitations, and limited familiarity with alternative models. Numerous advances over the past two decades have substantially reduced the inadequacies of science, data, and computing capability.

Among the policy issues debated, several were related to the concept of NER, including the fundamental definitions of the *Federal interest* in ecosystem restoration. There was considerable debate as to whether two categories of *motivation* for ecosystem restoration have emerged, and if so, the implications for specifying *Federal interest* in ecosystem restoration, *characterizing resources of significance*, *formulating objectives, selecting plan formulation and evaluation models,* and *justifying* proposed investments. These categories include 1) restoring the Nation's ecosystems to a "*more natural condition"-- independent of the significance of any specified resources and service flows*; and, 2) *restoring significant ecosystem resources* to a less degraded condition *as determined by services that flow from the resources*.

The notion of "significance", which plays an important role in ecosystem restoration planning was substantially discussed. The study concludes that the notion of "biodiversity associated with scarce species" (as defined by uniqueness and vulnerability), could be pursued to develop a "standard-measure" of "resource significance" that would help discriminate among NER investment choices. This notion can be distinguished from the fundamental notion of biodiversity in that it focuses on those species, communities, guilds and ecosystems designated to be of *ecological significance* by science-based reports, and the work of the WWF, TNC, and others. Pursuing this measure would be compatible with the habitat-based emphasis of the current Corps policy, and with the policy emphasis on resource scarcity as an indicator of significance. The standard units (see discussion in Section 3) would be based on characteristics of vulnerability and uniqueness, using methods developed by conservation biologists, and taking into account global rather than only localized significance. For example, while some significance may be inferred by plans supporting the North American Waterfowl Management Plan, greater significance would be attributed to plans that support a species such as black ducks – which are rare, relative to mallards – a specie included in the plan but not rare or vulnerable.

Such “scarce biodiversity” may not be the only measure of resource significance that contributes to NER, but placing emphasis and priority on such outputs is supportable because the recovery and protection of scarce resources determines the limits of future management options, including restoration options. If this approach proves applicable, recommendations for future restoration proposals that do note emphasize significant improvement in the status and sustainability of nationally scarce biodiversity, could be questioned as to their value as ecosystem restoration investments.

The study also concludes that a variety of existing ecological models are useful in formulating and justifying ecosystem restoration investments, contributing information to both forecasting ecosystem conditions, and specific outcomes related to resources of significance. The models can be usefully applied alone or in combination, depending upon the circumstances.

In the near term, a combination of community-habitat index models that forecast naturalness (including those such as IBI), and species-habitat index models that forecast suitability of the more natural state for the resources of significance can provide a basis for evaluating plan effects. In those instances where the more natural condition in itself is identified as the resource of social significance, ecosystem-level biodiversity models that are habitat based (e.g. IBI, WCHE, HGM) may serve satisfactorily once calibrated.

This conclusion does not, however, address the limitation that *habitat-based indicators of NER benefit are unlikely to capture all of the Federal interest* affected by restoration plans (as noted by the NRC). Other models, such as *functional capacity indices and process simulation models* are applicable for the multi-output analysis of benefits that appears to be required for multipurpose planning. Ecosystem process models have the advantage of generating more theoretically defensible and explicit results unsurpassed for communication and adaptive management, but are more costly. All existing models have shortcomings requiring substantial development effort, but especially so for the process simulation models. In addition, relatively few species-habitat models have been specifically developed for rare resources.

Additionally, *species-habitat index models usually have limitations, when used alone*, which make them less useful than alternative approaches. Ecosystem restoration planning models often need to account for at least two ecological indicators of importance, one that indicates the more natural support condition, and one or more that indicate condition of the dependent significant resources. A more natural, self-regulating condition is stipulated in Corps policy because the long-term maintenance of all resources of significance is most often assured by restoring the integrity of the support ecosystem. The single-species models provide a single index of relative environmental benefit based on the optimality of habitat for individual species, but are unreliable indicators of the more natural condition.

More recently developed *community-habitat indexes* set the optimum condition in the most natural ecosystem state and thereby provide a better alternative for indicating naturalness. However, when the resources of significance are identified independently of a more natural condition, it is more appropriate to use models that generate more than one output or a compatible combination of single-output models. Even then, all of the Federal interests may not be captured without additional indicators. Models such as the HGM functional capacity indexes and process simulation models are most suitable at that level of comprehensiveness. The explicitness of process simulation models outputs have advantages over the less explicit outputs of index models and can be particularly useful in NER and NED tradeoff analyses in search of an optimum combination. However, these models are among the least widely developed for restoration needs.

The study also concludes that significant technical obstacles preclude economic valuation of all possible restoration outcomes that could be evaluated in monetary terms. Furthermore, whether or not the utilitarian concept of economic value is the appropriate standard of “value” for evaluating restoration outcomes is open to question. Economic value may not indicate everything that stakeholders need to know about the desirability of restoration projects. This suggests that the current policy guidance that recognizes non-monetary NER outcomes as a category of effects separate from monetary effects is appropriate for evaluating restoration projects. However, a greater level of policy clarity is probably needed to help planners determine the appropriate restoration objectives and valuation standards for restoration planning.

The use of evaluation criteria that includes both non-monetary and monetary effects does not reduce the need for efficiency analysis in the NER planning context, and this need is recognized by Corps guidance. The cost-effectiveness analytical framework for single-purpose NER planning is very useful for evaluating the opportunity costs and marginal tradeoffs among alternative plans. That framework, which is essentially equivalent to the old P&S efficiency framework that plotted net NED effects against some measure of environmental quality change, is also applicable to multipurpose NED/NER planning, and can be readily extended to a multiple criteria efficiency analysis when NER outputs are best expressed in multiple, non-commensurate metrics.

The cost effectiveness framework is less discriminating as the number of choice criteria increases, making identification of more inclusive metrics an important pursuit. A focus for improving ecosystem restoration benefits analysis in the near term is to identify the monetary and non-monetary indicators of output needed to capture all significant effects, and ultimately to reduce them down to the minimum achievable.

## 7.2 Strategies.

The state of restoration planning capabilities, methods and models summarized above, resulted in a multi-component, three-stage strategy for improving environment benefits analysis, offered here for further consideration. The strategy addresses better use, refinement and further development of ecological assessment models, and improvement of staff understanding and application of assessment and evaluation tools. It also addresses the need for Corps policy and planning guidance to more carefully integrate ecological concepts, along with recent practical experiences in ecosystem restoration planning. The proposed strategy involves overlapping (I) *near*, *(II) intermediate*, and *(III) long-term* components, which can all start about the same time but differ with respect to the time of anticipated results. While the ideas below focus primarily on Corps specific actions, the need for collaboration with work going on in other agencies is emphasized.

The lack of appreciation for the *linkages among planning objectives, desired restoration outcomes, and model selection and use* appears to be at the root of some environmental benefits challenges. Establishing these linkages is fundamental to environmental benefits analysis, and potentially at the root of not only issues in model selection, but also some of the problems associated with alternative formulation, and project justification.

To the extent possible, the Corps should pursue the environmental benefits analysis improvement strategy in conjunction with other Federal and state agencies that can contribute to and benefit from these efforts. Shared development of methods for environmental benefits analysis might be expected to facilitate more compatible planning standards and practices across agencies.

**I.** The *near-term* or **Incremental** stage, from immediately to about 2 years, addresses the requirements of the current Corps planning regulations, seeks modest advances in improving environmental models, and emphasizes improving staff model selection and application capabilities relative to existing ecological models. Broadening this base of understanding and proficiency in selecting and applying existing models will provide the essential foundation for being able to apply new models as they are developed, in addition to improving environmental benefits analysis now.

**Ia. Models and Methods.** Modest model improvements could be made by moving from reliance on single-species index models, to greater use of community-based index models, either alone or in combination with single-species index models. Application improvements would emphasize linking project planning and ecosystem management goals in plan formulation.

A broad suite of existing and emerging models are available for use depending upon the type of project, system and scale of analysis. Few types of ecological models were developed specifically for restoration purposes and none are ideal, but some are more useful for forecasting ecosystem outputs. The examination of existing models concluded:

- Species-habitat models are sensitive to significant effects at the species level, but are not inclusive enough to formulate for restored natural ecosystem integrity.
- Community-habitat models are inclusive enough to formulate for more natural ecosystem integrity, but may be insensitive to significant effects at the species level
- Index models (e.g., HEP/HSI, IBI, HGM) are most widely available, but tend to exclude important systems context, require greater planner and stakeholder interpretation, and may require both community and species level index models for analysis.
- Process simulation models (e.g., ATLSS, CASM) are less available, but more output and process explicit, can incorporate complete systems contexts, can provide simultaneous output for conditions of naturalness and significant resources, and are superior for organizing lessons learned into improved model structure.
- As ecosystem planning conditions grow more complicated and the science improves, the advantages of process simulation models outweigh the expediency and lower-cost advantages of index models.

Future efforts should investigate the development of a metric based on the biodiversity of scarce species, and its usefulness in determining the significance of forecasted NER plan contributions to significant resources.

Models with the longest history of Corps use are the single-species habitat suitability indices (HSI models), originally developed for mitigation analysis before there was a Corps ecosystem restoration purpose and NER objective. In addition to the previously mentioned NRC and other comments about the shortcomings of using these models, the views of Corps staff vary regarding the adequacy of HSI/HEP models. For example:

- *They work, nothing else needed;*
- *Improvement is needed*;
- *HSIs are useful, but often there is not much underlying rationale or justification for the species and values selected – criteria are not clearly established. Differences in "with" and "without project" values are hard to justify and support;*
- *They are just a means to an end; used because they are easy and you have to do something for project justification*
- *HSIs are not a direct measure of output—suggest a weighted usable area as a more meaningful output measure to be derived from the HSI for the selected species and life history function.*

Some staff recommendations supported future work on developing process models and improved ways for conveying model results and associated information to non-technical decision-makers and stakeholders. Caution to avoid reinventing models that already exist was emphasized, as well as the need to retain flexibility in choices at the district level. The need to think ahead to consideration of outputs in tradeoff evaluation was also noted.

Ongoing efforts within the EMRRP[5] program, such as the development of templates for community-index models should contribute to model improvements in the near term.

**Ib. Capability in Model Application.** The immediate improvements can be made to environmental benefits analysis by improving the current understanding and ability to apply existing species- and community-index models separately and in combination. As noted earlier, this *broader base of understanding and proficiency will not only improve the current analysis*, but also *establish an essential foundation* for being able to apply new models as they are developed. Immediate analytical improvements can also be made by emphasizing the need to relate restoration objectives and outputs with model selection. The field identified a need for a “toolbox” of environmental evaluation models, and in some instances, the need for model selection and application instruction. The Planning Model Improvement Program Task Force also recommended a toolbox for planning models. Several efforts are underway that contribute to addressing these needs.

A protocol for selecting models for use in ecosystem restoration planning is being developed as a “Model Selection Reference Document”. This information will aid in the identification and selection of appropriate environmental models and methods that are currently available for use in ecosystem restoration planning. The Model Selection Reference Document is intended to be an optional resource or planning aide, rather than a set of “requirements”, as the field emphasized the need to retain flexibility in model selection. It summarizes different model types, attributes, and limitations, and infuses consideration of the broader Corps planning process -- emphasizing that model selection cannot be approached in isolation from the planning process as a whole. As such, the reference is structured along the Corps six step planning process. The document will serve to help:

1. Conceptualize the appropriate focus for quantitative assessment of environmental outcomes
2. Examine criteria for selecting model types based on the complexity of objectives and risks associated with proposed projects.
3. Identify, modify and develop appropriate assessment models
4. Use quantitative assessment results in plan evaluation and comparison.

The development of the Model Selection Reference Document by IWR staff includes the careful review and commentary of several Corps Planning Improvement Program and PROSPECT course instructors from ERDC-EL and NAE. This interaction between authors and instructors is essential to help assure consistency in course instruction material refinement and presentation, and broader infusion of the material, as appropriate, into existing and new training opportunities.

ERDC-EL is developing a web-based tool catalog as part of the SMART[6] R&D program. This effort and the Model Selection Reference are likely to be linked within the web-based EMRIS system, assuming sustained funding support for the efforts. Such efforts will provide a foundation for the “toolbox” requested by field staff.

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[5] Ecosystem Management and Restoration Research Program (EMRRP)
[6] System Wide Modeling, Assessment, and Restoration Technology (SMART)

<ins>More general ecosystem restoration planning capabilities</ins>. A number of training and other capability improvement opportunities exist to help bridge the gaps that presently exist many studies such as relating planning objectives, desired restoration outcomes, and model selection.

Discussion of environmental benefit analysis concepts and approaches also needs to be incorporated into a number of courses, workshops and other forums. The courses in the new Planner Core Curriculum, as well as nearly a dozen PROSPECT courses should be targeted to incorporate new analytical concepts and tools relevant to environmental benefits analysis and other aspects of ecosystem restoration planning, at appropriate levels of detail, depending upon the purpose and nature of the course.

Inserting material into the new Environmental Course within the Planner Corps Curriculum with the intent that this course will address model selection and application knowledge needs is not sufficient to address these needs. The treatment of models is only a small portion of the course, which is intended to cover nearly "everything environmental", including NEPA assessment and compliance with various other environmental laws. For some staff, a more in-depth treatment of application of the reference protocol would be helpful. Nearly all planners will need a better understanding of the use of model output information in the context of formulation and evaluation.

Additionally, in the short run, it may be useful to hold specialized workshops on model selection using the reference protocol, and actual district studies. Such workshops would improve district staff capabilities, assist the study, refine the instruction material for use in future courses and workshops, as well as advance the understanding of existing model application potential and future model development needs. Including staff from the stakeholder agencies in these workshops could also be beneficial.

**Ic. Policy and Guidance.** The need to link model selection with restoration objectives and desired outputs emphasizes that future policy development may need to refine or add explicit consideration of the notions of *significant resources, ecosystem integrity, ecosystem services, naturalness, self-regulation, resilience, stability, sustainability, production, materials cycling,* and other ideas. While some of these concepts have been more thoroughly developed than others, and many questions remain about concept validity and practical application, they can form a theoretical basis for NER evaluation. Additional discussion follows.

<ins>Restoration objectives and motives and ecological concepts.</ins> Corps policy regarding ecosystem restoration has evolved over the last decade and continues to do so. The currently stated Federal objective in ecosystem restoration is to increase the net quantity and/or quality of desired resources through the restoration of significant ecosystem function, structure and dynamic processes that have been degraded. Two possible motives for pursing restoration may be emerging, based on the accumulating experience with ecosystem restoration projects in the Corps. The first may be to secure a beneficial mix of ecosystem services that are more aligned with natural ecosystem parts

and processes. A second may be to restore the "naturalness" of ecosystem properties *as end in itself*, independent of the resulting mix of services and benefits. .

Current Corps guidance does not specifically identify the desired ends of restoration as naturalness for its own sake. Instead, ecosystem restoration guidance emphasizes the "significance" of resources and restoration effects for guiding and justifying restoration while establishing restoration of more natural ecosystem structure and function as the preferred condition for supporting significant resources and natural services. The significance concept as defined by Corps guidance seems broad enough to encompass both naturalness and associated services as desired restoration ends.

The metrics and associated methods used for evaluating restoration projects outputs should follow from the desired ends of restoration in any particular context. If restoration of hydrology and geomorphology represents one valued end to project stakeholders, then the non-monetary metrics chosen to characterize and evaluate project effects might be derived from the pre-disturbance hydrology and geomorphology, or some other relevant reference condition. On the other hand, if the restoration of natural ecosystem services is of prime concern, then project evaluation requires moving beyond metrics indicating a more natural state to include metrics that indicate the desired direction of change in desired service outcomes.

NER Evaluation framework. Corps rules do not require the monetary valuation of restoration outputs, or the use of a monetary standard to identify and choose among economically efficient plans. CE/ICA is used to help assure cost effectiveness in achieving different levels restoration output and to subjectively determine what level of restoration output is worth the cost to achieve it.

This CE/ICA framework is most useful when restoration outputs can be adequately characterized in terms of a single non-monetary output metric. But in many restoration contexts it may not be reasonable or possible to characterize and evaluate outputs in terms of a single metric. In that case, the two-dimensional CE framework can be readily extended to an efficiency analysis defined over multiple criteria. For example, in a case in which plans are evaluated in terms of cost and two non-commensurable, non-monetary measures of NER output, the efficiency analysis would identify plans for which more of one NER output could not be obtained through choice of another plan without incurring higher costs or obtaining less of the other NER output. Additional guidance or training on evaluation under these circumstances may be helpful.

The Corps recently published interim guidance for the evaluation of multipurpose NED/NER plans (EC 1105-2-404). As restoration policy evolves, giving consideration to the concepts noted above, it will be necessary to assure that the evolution of this guidance is consistent with the evolution of restoration policy, along with insights gained from practical application of the EC.

Policy studies on NER and ecosystem services will contribute insights on the above issues. Two policy studies initiated in FY 03 are examining the concept of NER, and the concept of ecosystem services for potential application in Civil Works ecosystem restoration planning. In the first, the NER concept is being examined as a federal objective and basis for formulating ecosystem restoration projects. Ecosystem services is being examined for potential usefulness in ecosystem restoration planning, particularly in the context of joint projects with both NED and NER outputs.

**II.** The *intermediate* or **Next Generation** stage, from immediately to about five years, would pursue a fundamental rethinking of the NER objective and desired outputs. Specifically, it would more intensely pursue the idea that ecosystems provide important mixes of ecological services, and the possible advantages and practicality of defining an NER account that specifies these services (both monetary and non-monetary). Further, it would seek to improve the ability to evaluate specified services though the use of ecosystem process simulation models at proper landscape scales. New analytical frameworks for multipurpose NED/NER planning would be explored, including the opportunity cost framework recommended by the *Principles and Standards* several decades ago for evaluating tradeoffs between plan economic and non-monetary environmental effects.

**IIa. Models and Methods.** During this stage, the development and refinement of ecosystem process models that estimate actual outputs would be emphasized. Efforts to develop and refine ecological models for environmental benefits analysis should be integrally linked to economic and decision making frameworks. This linkage is essential to help ensure that the models and results adequately fit the evaluation frameworks used in Corps planning, and to inform the further evolution of those frameworks.

Research programs such as the EMRRP, SMART[7] and TOWNS[8], and others, along with the EMRIS[9] system could play a central role in the development of guidance for using existing ecological models, expansion of existing prototypes to new applications, and development of new models. Efforts should begin immediately to strategically refine and merge the need for this effort into ongoing and planned research.

Within the EMRRP, work proposed to begin in FY'04 would develop a framework that links habitat analysis, dynamic process modeling, and spatial statistics for application in aquatic systems. The work description says that products will incorporate contemporary ecological principles and, current techniques, lend to adaptation and enhancement as new tools are developed and new ecosystem principles unfold. "Tools developed under this work unit will allow Districts to assess and quantify the impacts and benefits from a wide range of water resource projects while maintaining flexibility so that the analysis procedure is appropriate to the project needs and constraints."

The areas of focus within SMART that seem to have potential for this include: Environmental Processes and Resource Responses; Environmental Assessment and

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[7] System-wide Modeling, Assessment and Restoration Technologies (SMART)
[8] Technologies and Operational Innovations for Urban Watershed Networks (TOWNS)
[9] Ecosystem Management and Restoration Information System (EMRIS)

Prediction Technologies; Decision Support and Application Technologies. Among ongoing or planned efforts is a compilation of ecological tools and approaches for system-wide assessments, including habitat-index models, empirical (e.g., statistical) numerical (e.g., process simulation) models, and geospatial techniques (e.g., GIS). Plans include making them available via a user-friendly, web-based framework with a decision support system to facilitate effective selection of assessment tools. Linkages to economic evaluation frameworks will be essential, and this should occur integrally, rather than sequentially. There are also plans to develop prototype applications of system-wide assessment frameworks by working with districts and their partners to develop conceptual models for implementation in project management plans and feasibility studies.

Within the TOWNS R&D program, work proposed on the value of evacuated floodplains could contribute to improving environmental benefits analysis. The work, if funded, would examine alternative uses for, and valuation approaches and measures for evacuated floodplains.

Potential applications of the Ecosystem Functions Model (EFM) beyond the Sacramento-San Joaquin basin should be explored[10]. The Watershed Analysis Tool (WAT), being developed as part of the Flood and Coastal Systems R&D Initiative is integrating HEC NexGen software for watershed studies. Products will streamline the analytical and reporting processes of the NexGen software, while producing more consistent results for watershed-type studies. WAT will link to data processing and modeling and spatially referenced displays, as well as to other models, including to EFM.

The potential roles for and contributions from the Environmental Modeling and System-wide Assessment Center (EMSAC), recently formed within ERDC, should also be explored. The EMSAC is chartered to enhance coordination and technical focus for modeling (assessment and forecasting) activities in order to advance system-wide applications of predictive environmental modeling, assessment, and management tools. It uses a matrix of ERDC elements to form technical teams of engineers and scientists to solve complex system-wide environmental problems involving complex environmental systems across multiple media and over broad spatial scales. The EMSAC integrates R&D in hydrodynamics, hydrology, ecology, and related disciplines, along with applications of technology, modeling and informatics for alternatives analysis and decision-making.

**IIb. Capability in Model Application.** Improvements in model use and the application of model output information in investment and management decision making could be facilitated by the formation of *model application assistance teams*. Such assistance, applied in conjunction with multi-agency workshops targeted toward actual projects,

[10] EFM uses statistical indicators to link hydrologic regime to aspects of the ecosystem (plant community and fish community). Indicators are tested under different flow regimes for with and without project conditions, and results help users to identify the direction of change (improve, no change, or decline) for the individual ecological parameters. Results can be expressed as spatial areas which can be used in incremental cost analyses.

could be useful in fostering model use capabilities, innovation, and understanding – both assisting a given study, and advancing the state of the science in model development and application.

**IIc. Policy and Guidance.** Efforts during this stage would pursue further refinement of the NER concept and outputs, relative to ecosystem goods and services, along with alterative analytical frameworks useful in Corps planning, especially for joint NED/NER projects. Emphasis would be placed on conducting ecological analysis in a hierarchical fashion to better serve overall ecosystem management goals. Appropriate landscape and scale effects and considerations (river basin, watershed, flood plain) would be discussed for all projects, providing an improved context for the significance of restoration outputs.

Concept of NER. The understanding of the concept of NER purpose and the NER plan is thought to be clear to some Corps staff, but often not to others. For example, some stated:

- *A general discomfort with justification policy for NER plans, especially in joint formulation*
- *Confusion regarding whether or not restoration pertained to "degradation" "caused by natural change"*
- *General uncertainty about how to determine when an NER project was not justified.*

A policy study started in FY03 has begun a more critical examination of the NER concept as a federal objective and the basis for formulating ecosystem restoration projects. The NER study will examine the potential usefulness of the concept of ecosystem services for defining NER as a formulation construct and for developing a set of standard methods and metrics for characterizing and evaluating NER outputs.

Ecosystem Services. At any given time, the structural features and ecological processes of an ecosystem[11] yield a mix of functions that in turn provide services valued by society. These include both natural and humanly enhanced services. Natural ecosystem services have been defined as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Dailey, 1997). Corps authorities to pursue ecosystem restoration reflect increased public recognition and appreciation of the contribution to human welfare provided by ecosystem services.

Corps guidance (ER 1105-2-100) directs planners to habitat services, which comprise only a subset of the broader suite of ecosystem services of interest to society. Equating biological *resources* with *ecosystem resources* limits evaluation perspective. This limitation in turn reinforces the use of HEP and similar design tools that address only part of the comprehensive ecosystem restoration emphasized as the proper approach to objective setting in various NRC reports. The NRC concluded: "The difficulty with HEP and similar methods is that they capture only a part of the national interest" (NRC 1999).

[11] --as affected by environmental forces and constraints, management actions, and social and economic activity in the area--

The understanding and perceived potential value of the concept of ecosystem services in water resources planning varies across Corps staff. With regard to pursuing further understanding and application of the concept of ecosystem services, some staff say:

- *Try it – often sponsor interest isn't habitat improvement per se, but improved water quantity or quality as restoration outcome*
- *Recognizing and "legitimatizing" other benefits would improve our analysis*
- *National values are questionable; Would the list of services be national or developed on a case-by-case basis?*
- *General list could be difficult to produce, except maybe in broad categories of functions. Still, it might help to create such a thing as part of the planning process, at least at the project level.*
- *Don't need to do this.*
- *Could be useful for combined NED/NER plans.*
- *Could help in determining "is it worth it?"*

Depending upon whether the current support for integrated formulation persists, reintroduction of the NED-EQ tradeoffs, and return to P&S multi-objective formulation and evaluation procedures may be pursued as a means to further support the elements of the **sustainability** philosophy expressed in the PCSD (1996), and evolving through implementation of the Corps' Environmental Operating Principles.

The broader notion of environmental analysis may integrate the "NEPA process" into the P&G/P&S planning process, thus eliminating differing standards and principles for evaluation for ecosystem restoration planning and environmental impact assessment. Potential changes needed in policy and guidance would be identified.

**III.** Over the *longer-term*, from immediately to about ten years, efforts would be made to pursue the economic valuation of ecosystem services. The objective of this **Monetization** stage would be to marry ecological process simulation models with economic valuation methods towards more comprehensive evaluation of restoration outcomes in economic terms. If deemed practical and acceptable, this could lead to the development of standard analytical tools for different ecosystem services to mirror the techniques for evaluating NED outputs specified by the P&G.

The field and other staff have expressed mixed feelings about pursuing full monetization. Among the various views are:

- *Let's try it*
- *It's a bad idea*
- *Too expensive, there is no confidence in results*
- *Explore it but don't <u>require</u> it, especially for CAP*
- *See work done by NOAA, Forest Service and universities*
- *There could be potential impacts on Regulatory and would we monetize endangered species habitat?*
- *Different regions have different needs*
- *Perhaps it could be considered in terms of "replacement costs" – (e.g. wetland bio-filtration vs. a treatment plant"*

In general, the economic techniques outlined in the P&G for valuing traditional civil works outputs in monetary terms are also generally applicable to the types of "natural" ecosystem service outputs likely to be associated with ecosystem restoration. However, there are considerable technical obstacles to comprehensive monetary accounting of restoration benefits. *Non-economic obstacles* relate to the complex biological linkages between restoration actions and service outcomes that are often not well understood and readily predictable. *Economic obstacles* relate to methodological limitations for measuring non-market benefits of service outcomes that affect the quality of human life in ways that have no close connection to the use of marketed goods.

In addition to these technical obstacles, some economists, political scientists and philosophers have questioned the relevance of the economic concept of value with respect to ecosystem services such as the sustenance of endangered species that may often be the focus of restoration. Challenges from these critics could hinder the political acceptability of adopting a monetary standard for evaluating and justifying restoration projects.

Nevertheless, in some cases it should be technically possible and practical to estimate monetary values for restoration effects that could be used to inform decisions in ways that are politically acceptable. An obvious example is when restoration project plans measurably affect traditional NED outputs such as flood regulation. In such cases, these effects should be valued and used within the CE/IC framework for evaluating and comparing plan alternatives.

Several efforts ongoing within the Decision Methodologies Research Program will contribute to this pursuit. These include identification of recent and ongoing district studies that monetized environmental outputs, identification of examples in other agencies, and a literature review. In addition, a test case has been proposed that would apply monetization to a completed ecosystem restoration project, in an effort to examine whether and how this information could have been useful in decision making. Other IWR research is examining the potential use of air quality benefits, from reduced emissions attributed to inland waterway shipping relative to truck or rail modes of transportation.

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