



**US Army Corps
of Engineers®**
Engineer Research and
Development Center

ERDC
INNOVATIVE SOLUTIONS
for a safer, better world

Ecosystem Management and Restoration Research Program

Measuring Environmental Value in Nonmonetary Terms: A Review of Common Practices and Elements

Richard A. Cole

May 2014

The US Army Engineer Research and Development Center (ERDC) solves the nation's toughest engineering and environmental challenges. ERDC develops innovative solutions in civil and military engineering, geospatial sciences, water resources, and environmental sciences for the Army, the Department of Defense, civilian agencies, and our nation's public good. Find out more at www.erdcl.usace.army.mil.

To search for other technical reports published by ERDC, visit the ERDC online library at <http://acwc.sdp.sirsi.net/client/default>.

Measuring Environmental Value in Nonmonetary Terms: A Review of Common Practices and Elements

Richard A. Cole

*Institute for Water Resources
United States Army Corps of Engineers
7701 Telegraph Road, Casey Building
Alexandria VA 22315-3868*

Final report

Approved for public release; distribution is unlimited.

Prepared for US Army Corps of Engineers
Washington, DC 20314-1000

Monitored by US Army Research and Development Center
3909 Halls Ferry Road, Vicksburg, MS 39180-6199

Abstract

This review was undertaken to address concerns raised by the US Army Corps of Engineers (Corps) regarding the value of projects authorized to improve environmental quality. Value gained from present resource use can typically be measured in monetary terms. More controversial -- and prohibited by Corps policy -- is the monetary measurement of the nonuse value gained by deferring present-day use in favor of leaving a heritage for future generations. Environmental value is commonly indicated by the objectives of government legislation and by non-government mission statements and bylaws. The value added by objective achievement is indicated by many different, incomparable metrics, which often do not differentiate use value from nonuse value. The Corps is an exception because federal policy requires water resources agencies to quantify benefits and costs in monetary terms when acceptable and in non-monetary terms when monetary measurement is not acceptable. This includes heritage value recognized as important by the Corps in key environmental legislation, and by certain conservation NGOs. Key elements of natural heritage value include resource security from extinction, resource distinctiveness, risk of investment failure, and costs. These elements may provide a basis for comprehensively indicating the value added by ecosystem restoration done by the Corps.

DISCLAIMER: The contents of this report are not to be used for advertising, publication, or promotional purposes. Citation of trade names does not constitute an official endorsement or approval of the use of such commercial products. All product names and trademarks cited are the property of their respective owners. The findings of this report are not to be construed as an official Department of the Army position unless so designated by other authorized documents.

DESTROY THIS REPORT WHEN NO LONGER NEEDED. DO NOT RETURN IT TO THE ORIGINATOR.

Contents

Figures and Tables.....	iv
Preface.....	v
1 Introduction.....	1
Review Purpose	1
Why Nonmonetary Measures of Value?	2
2 Methods	6
3 Values Classification	7
Instrumental Value	8
Use Value.....	9
Valuation of Direct and Indirect Use	10
Nonuse Value	10
Non-instrumental Value	12
4 Measuring Non-monetary Value in US Federal Agencies.....	15
Public Goals Achievement, Cost-Effectiveness, and Public Benefit	15
Public Goals and Budget Limitations.....	15
Cost-Effectiveness.....	17
No Common Metric Despite a Common Goal.....	18
Programmatic Authorities	22
Regulatory Programs	22
Project Authorities	33
Environmental Protection	33
EQ Improvement Using an Ecosystem Restoration Approach.....	36
5 Measuring Nonmonetary Value Outside Federal Agencies.....	45
Other Government Agencies.....	45
Professional Scientific Organizations.....	45
Regulatory NGOs	46
Conservancies	46
Conservancy Values	47
Conservancy Planning.....	49
Conservancy Objectives.....	51
Conservancy Strategies	55
Conservancy Effectiveness.....	59
6 Conclusions.....	62
References	64
Report Documentation Page	

Figures

Figures

Figure 1. A classification of value based on utility, use and nonuse, and intrinsic value (created from information provided by NRC 2005).	8
Figure 2. In the federal government process, public desires are translated into statutory goals. Agency authority establishes objectives and policies that promote goal achievement. Achievement may be measured qualitatively or quantitatively by performance indicators (e.g., acres protected, number of endangered species down-listed). More important is the public sense of improved welfare, which may lead to statute amendment or repeal, or more or less annual funding of the agency program.	16
Figure 3. Example performance indicators for environmental monitoring.....	21
Figure 4. This schematic for the ecosystem restoration concept of the Corps illustrates some of the primary considerations. Based on its authorities, ecosystem restoration takes place through restoration of the geophysical environment—hydrology and associated geomorphology—in the proper ecosystem context and connectivity to assure that both the biologically desirable species and their supporting ecosystem are restored for their direct and indirect nonuse value.	41

Preface

This technical report is the first in a series of three reports about the same topic: measuring the benefits of environmental quality restoration activities performed by the US Army Corps of Engineers. The author of the report, Dr. Richard A. Cole, is an environmental planner at the Institute for Water Resources, US Army Corps of Engineers (Robert Pietrowsky, Director). For their report review comments and discussions, the author is indebted to Ellen Cummings from USACE Headquarters; to Lillian Almodovar, Michael Lee, and Dr. Norman Starler from the Institute for Water Resources; and to Drs. Andrew Casper and Barry Payne from the Engineer Research and Development Center, US Army Corps of Engineers. The author especially benefited from discussions with Bruce Carlson, Lynn Martin, Paul Scodari, Leigh Skaggs and Dr. Paul Wagner.

This report is published as a product of the Ecosystem Management and Restoration Research Program (EMRRP) at the US Army Research and Development Center (ERDC), Vicksburg, MS. Glenn Rhett was Program Manager, Al Cofrancesco was Technical Director.

During the conduct of this work, Dr. Elizabeth Felming was Director of the Environmental Laboratory (EL), COL Kevin J. Wilson was Commander and Executive Director of ERDC and Dr. Jeffery P. Holland was Director.

1 Introduction

Review Purpose

The purpose of this report is to review both qualitative and quantitative nonmonetary measurement practices for environmental value in government and nongovernment organizations (NGOs), and to reveal any common elements. The review was done in large part to inform the development of a new nonmonetary metric to measure environmental quality (EQ) protection and improvement benefits. The United States Army Corps of Engineers (Corps) has a need for such standards of measurement in its Civil Works project and program planning. Corps project planning guidance (USACE 2000) requires use of nonmonetary metrics to quantify “net contributions to increases in ecosystem value” as stated in the federal planning objective (USACE 2000, pg 2-2). This is consistent with the requirement that all value that can be measured in monetary terms be so measured as contributions to net economic development (NED) value. Economic development projects must be authorized for purposes other than the ecosystem restoration purpose. The results of the review in general are relevant to researchers, managers and decision-makers interested in environmental decision process in government settings.

It is not the intent of this review to describe in detail the various ecological metrics that have been used to indicate project and program performance in the Corps and elsewhere; this has been done by others (e.g., Bartoldus 1997, Stakhiv et al. 2003, O’Neil et al. 2001, NRC 2000, The H. John Heinz III Center for Science, Economics and the Environment 2008). Neither is it the intent of this review to develop a new metric. That is treated elsewhere (Cole 2010). This review does not thoroughly assess the new metrics applicability in project and program planning; that will be addressed in future analyses and publications (Cole 2011). The intent of this technical report is to review practices for those elements that capture a significant amount of the value added that is not, by policy, acceptably measured in monetary terms.

The review starts with a brief, introductory history that explains why nonmonetary measurement of environmental value is important. That is followed by a classification of values and how their measurement is approached by economists. It then describes the alternative approaches that

have been taken by agencies and NGOs to measure values when monetization was either ignored or viewed to be unacceptable. The last section summarizes the most commonly used criteria for indicating the ecosystem-associated value of EQ in nonmonetary terms.

Why Nonmonetary Measures of Value?

The Corps did not independently elect to use nonmonetary measures of environmental value. The approach had its genesis in the form of guidance that was issued in response to laws passed in the 1960s and 1970s. These laws were enacted in response to the public's demand for full consideration of environmental value (Sale 1993, Kline 1997, Hays 1998 and 2000). This consideration included the value of EQ that was excluded from monetary expression and typically ignored in benefits and cost analysis. Also prominently included in that consideration were effects on EQ that were not measured in monetary terms, because no widely acceptable way had been developed. The water resources development agencies were particularly criticized because they had justified water resources development for decades based on economic (monetary) criteria, as required by Congress, and dismissed development effects on environmental value (that economists did not ordinarily attempt to measure) (Hays 1987; Dunlap and Mertig 1992). Recreational quality of the outdoors environment fit into that category before acceptable economic methods were developed for its monetary measurement in the 1950s.

The requirement for benefit-cost analysis in water resources development planning grew out of concern that the most beneficial projects were not always receiving construction priority. The federal water resources development agencies were uniquely targeted in large part because project planning and construction investments were very large, comprising a fraction of the federal budget about two orders of magnitude larger than today and much larger than the budgets of other resource management agencies. The Corps' budget alone was nearly 5 percent of the total federal budget in 1936, when economic justification was first required.

A plan was rejected, in concept, if the estimated gross benefit minus cost (net benefit) was not positive. Ordinarily, the plan with the highest net benefit to the nation was the one recommended to Congress for funding, but the approach varied greatly from project to project. Typically, however, values were ignored if they could not be acceptably measured in dollars or were not protected by law. Until the late 1950s (when the Fish and Wildlife

Coordination Act was strengthened), effects on public outdoor recreation, environmental aesthetics, and natural and cultural heritage were among those considerations usually dismissed because there was no generally acceptable way to measure their value in monetary terms.

After World War II, the project planning and construction budgets began to draw the critical attention of resource development economists (Maass 1951) and advocates for outdoor recreation and other environmental concerns not captured in monetary measures of environmental value (Moore and Moore 1987; Hays 1987, 1998 and 2000). These concerns were often referred to as EQ and included certain health issues and the steep declines in the abundance of certain wild species both individually and in community collectives of natural ecosystems. This rapid growth in environmental interest ushered in an era of unprecedented federal environmental legislation.

After the National Environmental Policy Act was passed in 1969, other new environmental laws opened government agency performance to public scrutiny and to public law suits when agencies appeared too lax in executing and enforcing the laws. Continued public interest in the goals of environmental law, expressed primarily through environmental NGOs, was evidence of strong public demand for a less sullied and wilder outdoor environment, and a more complete natural heritage to bequeath to subsequent generations. Sectors of private industry and agriculture -- and even certain federal agencies -- bitterly fought aspects of the environmental movement because they thought the existing economic value of the environment, revealed in market prices, exceeded environmental value that could not be measured in monetary terms. However, public commitment to protecting environmental value for reasons other than economic ones prevailed because of broad public support (Hays 1987, Dunlap and Mertig 1992).

Many people believed the nation was duty-bound to refrain from some immediate resource use so that some natural environments would be left intact for future opportunities -- especially for outdoor recreation and education, but also to sustain rare species in their supporting ecosystems. That same sense of natural heritage had motivated in large part the setting aside of more public land in national parks and wilderness area at some cost to those who might use that land for conflicting purposes (e.g., logging, grazing, recreational housing). The heritage value of natural environments

also contributed to the motivation of private conservancies to invest in the protection of native biodiversity through property and easement purchase.

The concern for this natural heritage was explicit in the congressional objectives of certain environmental laws and treaties, which made clear that all significant environmental value, whether measurable in monetary terms or not, was to be considered in federal actions that affected the environment. NEPA holds beneficial use of the environment and preservation of natural and cultural heritage equally high among environmental policy goals. Economic development by federal agencies is consistent with environmental policy to the extent it contributes to beneficial use of the environment while assuring preservation of national heritage. A natural heritage motivation was in significant part behind the Endangered Species Act (ESA) as well. The United States recognized in the Convention on International Trade in Endangered Species of Wild Fauna and Flora that "...wild fauna and flora in their many beautiful and varied forms are an irreplaceable part of the natural systems of the earth which must be protected for this and the generations to come." The ESA was passed in large part to honor this and other international pledges. In effect, it established a federal policy to sustain the viability of plant and animal species threatened by human action (except for a few insect pests).

More specifically, the Water Resources Planning Act (WRPA) of 1965 requires consideration of all significant social effects of federal water resource development projects on resource value, whether or not they could be acceptably expressed in monetary terms. Early guidance developed for the WRPA indicated that along with aesthetic value, natural ecosystem viability and cultural heritage values were important motivations for protecting and improving the environment for value, although this value was not widely viewed as measurable in monetary terms. This value was called EQ, just as it was referred to in the goals of the WRPA. Section 206 of the Water Resources Development Act of 1996 directed aquatic ecosystem restoration done by the Corps to restore and protect ecosystems to improve EQ. Consistent with the ecosystem protection aspect of that authority, Corps policy guidance indicates that the value of ecological resources that justifies ecosystems restoration investment is to be measured in nonmonetary terms (USACE 2000 pg 2-1) and excludes cultural and aesthetic resources (USACE 2000).

The environmental movement spread to the international deliberations of the United Nations (the UN). The UN built on early concepts of sustainable development to establish a principle of international development based on sustaining essential aspects of the economy, society, and the environment (World Commission on Environment and Development 1987). Numerous studies have reviewed UN policies, concepts and applications, and reduced them to principles (e.g., Muschett 1997, NRC 1999a, Ekins 2000, Furtado et al. 2000). Unlike the federal guidance that resulted from implementing WRPA, which lumped cultural and natural resources into a concept of EQ, the separation of sustainable development into social (including cultural), environmental and economic aspects implicitly recognized inherent differences in the measurement of attributes and values. Economic value is measured in common units of exchange—monetary units. But social-cultural and natural environmental values that have not been acceptably monetized have been measured in myriad ways, complicating their understanding. To add to the complexity, values that could be acceptably monetized are often not monetized. These typically involve services that are not priced directly by the marketplace and must be priced indirectly through other means. Recreational services are common examples.

Many different incommensurate indicators of EQ have been used in government and NGOs alike, both within and among programs. The diversity of project and program performance indicators has confounded those who attempt to assess program performance. This diversity of indicators has also complicated coordination and collaboration across programs established to promote environmental sustainability and sustainable development.

2 Methods

Standard review methods were used, including a general literature search and the perusal of recent literature reviews for concept synthesis and for references to primary literature of particular relevance. Much use was made of recent literature reviews, especially for advances made in the field of conservation biology relevant to NGO investment strategies. Relevant information also was sought from Internet searches and government policy and authority reviews. The review was focused by the special needs of the Corps of Engineers in its plan formulation and evaluation, and in program budget planning, all of which are defined in Civil Works authorities and policies. Therefore, the review concentrated on information relevant to qualitative and quantitative measurement of ecological value and, most specifically, ecosystem value that cannot, according to Civil Works policy, be measured in monetary terms. For the most part, economic literature pertaining to monetary valuation methods was not included. The large body of detailed information pertaining to biological indices of water and other environmental condition was not reviewed except to determine how those measures relate in general to environmental value.

3 Values Classification

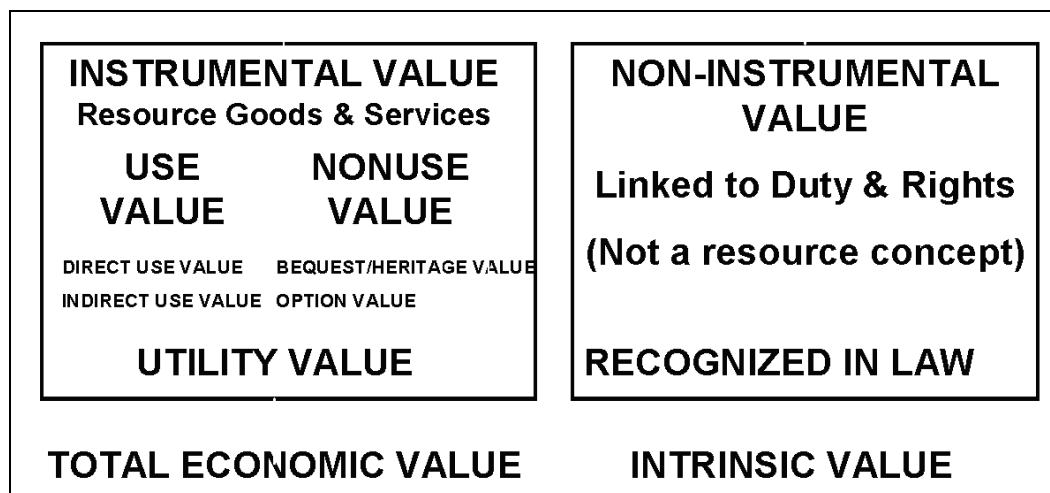
People value the environment for many reasons and individuals can hold multiple sets of values simultaneously (NRC 2005). The value described in this report is based on human awareness of physical, chemical, and biological diversity in the material world, the utility of that diversity (primarily the biological diversity), and its intrinsic value, if any. The value of utility is usually the extent to which material diversity does or may potentially serve as resources for human use. Values based in utility are directly compared and traded based on prices in the marketplace (market valuation), on estimates of what market prices would be if a market could be established (non-market valuation), and on physical or index indicators of value. Intrinsic value is independent of utility and lies entirely outside the limits of what can be traded and measured in monetary terms. Some utility is obvious because it is revealed in use behavior and is directly or indirectly measurable in monetary terms. Other utility is potential and can be monetarily valued by people only through their stated intent, the accuracy of which is questioned by numerous economists and decision-makers. This nonuse (or passive use) category is of particular relevance to anyone interested in negotiating for the protection and improvement of EQ that cannot be acceptably measured in monetary terms.

These and other categories of value have been distinguished and organized in various classification schemes, usually with the intent of organizing our existing understanding of economic valuation process and its limitations (Randall 1991, Freeman 2003, and NRC 2005). Values classification is useful for sorting through complex aggregates of values often claimed for EQ and for sorting out the types of values that can be negotiated in tradeoff settings and measured with confidence in monetary terms versus values that can be measured with much less confidence in monetary terms. A classification scheme can also reveal something about how direct and indirect motivations behind the goals of environmental law and organizational missions relate to value and its measurement.

The classification depicted in Figure 1 is based largely on whether any valuation process is acceptable based on economic criteria, which are limited to instrumental (or utilitarian) value and exclude noninstrumental (or intrinsic) value. It is based largely on reviews and concept development

in NRC (2005), Freeman (2003), and, specifically with respect to biodiversity, Barbier et al. (1995). Each category and subcategory of value is briefly described below.

Figure 1. A classification of value based on utility (use and nonuse) and intrinsic value (created from information provided by NRC 2005).



Instrumental Value

Instrumental value is based on resource utility that can be compared and traded according to some common understanding of exchange value, such as the prices charged in a market place. It can be measured in units of exchange or trade, albeit not necessarily in reliable monetary units. All instrumental value is tradable and theoretically measurable in comparable units of price. Setting prices in monetary terms for a wide variety of goods and services facilitates informed investment choices, especially in complex market settings. Many economists aspire to measure all instrumental value in monetary units for that reason. However, some instrumental values are more acceptably measured in monetary units than others. Examples of the instrumental value of water resources include the value of deep water for navigation, clean water for drinking, intact watersheds for clean water, water birds for watching, floodplains for human settlement, and natural areas set aside from use to sustain future opportunities. In this last example, the value of resource use that reduces future opportunities for use is traded for the value associated with sustaining future opportunities.

Use Value

Direct Use

All instrumental value is categorized with respect to immediate use or nonuse of resources and whether they are directly or indirectly used (Figure 1). The direct use of environmental goods and services by people is resource-interactive, whether it be consumptive use (e.g., commercial fish harvest, drinking water) or non-consumptive use (bird watching, navigation) (NRC 2005). Use value is revealed by past human behavior; most obviously, in the prices paid for common commodities and services. Individual preferences are demonstrated in the trades made for goods and services. The willingness to pay or to accept payment is determined by the demand for the good or service, income, and the availability of substitute goods or services. Some use value, including that of many public services, cannot be priced through the market, but can be priced indirectly through various techniques. Whereas some public goods and services can be priced and sold (e.g., park, rangeland, and navigation use fees) many more cannot because access to them cannot be controlled (NRC 2005). A variety of methods are used to reveal or impute values in monetary terms when there is no market for the goods and services, as is the case with many natural goods and services (e.g., Apogee Research, Inc.1996; Holl and Howarth 2000; King and Mazzota 2002, Freeman 2003).

The goods and services of natural environmental resources (e.g., land, water, air, and biological resources) are directly and beneficially used in many ways. People benefit from the many commodities produced from environmental resources including food, building materials and medication among others. They benefit from recreational and esthetic use of some natural environment resources more so than modified alternatives. They also benefit from the use of naturally clean air and water, which is less costly to use than artificially cleaned resources. In addition, they benefit from continuous use of land and water reliably free of extreme disease, flooding, drought, heat and other disturbance moderated by natural environmental structure and process. More detailed descriptions of natural services are provided by Daily (1997), Farber et al. (2006), and NRC (2005).

Indirect Use

The land, water and other environmental conditions needed to sustain directly used ecological resources derive value indirectly through the value

of the used resources. The value of floodplain land, for example, is derived indirectly from the direct use of the resources it supports, such as agricultural, recreational wildlife, transportation, and housing resources. The land and the ecosystem have no instrumental resource value independent of their direct use. In concept, land that cannot ever support agriculture, housing, roads, wildlife or other natural or developed human resources has no resource value, although it may be difficult to find examples. Similarly, the resource value of ecosystem restoration is no more than the value summed from each of the directly used goods and services provided by the increased quantity and quality of ecosystem support. The extent that habitat is improved (made more valuable) is indirectly determined by the value gained from the biological populations that inhabit it. It has no habitat value independent of the actual and potential inhabitants.

Valuation of Direct and Indirect Use

Among the methods used to estimate value, production-function methods that link environmental support resources to specific resource utility are especially relevant to valuation of goods and services produced by natural ecosystems. The resource outputs from environmental processes serve as inputs to directly used resources (Barbier 1994, 2000). For example, the recreational value of fish and wildlife depends on spatially explicit estimates of fish and wildlife production and stock abundance, which -- with measures of access -- determine recreational opportunity (e.g., Cole and Ward 1996). The underlying ecological processes in the supporting ecosystem that produces the recreational wildlife are valued indirectly from the estimated recreational value of the wildlife. All methods, including market pricing, involve some uncertainty in value estimation and that uncertainty increases dramatically as the time period included in the benefits estimation increases (e.g., Randall and Farmer 1995).

Nonuse Value

Nonuse value is associated with setting aside environmental resources from destructive use for various reasons independent of their present resource use (Freeman 2003). This is also called existence value, and it is derived from an individual's knowledge (or belief) that something exists independent of any present use. It has also been called *in absentia* value because use typically requires human presence (Pearsall 1984). Existence value is subdivided primarily in two subcategories: option and bequest value.

Option value is derived from managing the risk of losing something of value. Much of that value may not be independent of present use and is measurable in monetary terms (NRC 2005). The value qualifies as nonuse value when it is independent of present use. Krutilla (1967) was one of the first to recognize the value in preserving a species or a natural area to protect the options for their “mere existence.” This motivation appears to be important among some members of environmental NGOs. This is not to be confused with use value derived from managing the uncertainty in losing unknown economic value from ecosystems by sustaining all resource production potential, some of which might be revealed to have economic value. This latter motivation appears to be important among some tropical nations for setting aside rainforest from present use. All of the use and nonuse benefits associated with option value are realized by the individual who pays for it.

Krutilla (1967) also recognized the nonuse value in “bequeathing natural resources to one’s heirs.” This natural heritage value is a bequest value. Unlike option value, which is motivated by protection of the investor’s personal wealth, bequest value is altruistic; i.e., motivated by protection of the investor’s benefactors. Private bequests may be to individuals but public bequests are open to all. Natural heritage value derives from the desire to pass on intact resources to all citizens of future generations for the opportunity to use the resources, or not, as they desire. It may appear that the past investment in national parks and monuments and other set asides indicate bequest value, but government decisions to make these investments are not proof of value, since it may have been greatly under- or over-estimated. It also reflects the recreational use value of the set-asides. Freeman (2003) emphasizes the difficulties associated with separating use and nonuse values, and the need for more research in this area. It is also difficult to separate option value from bequest value, but because public policies often require many years to fully implement, option value is less likely to be realized because of policy implementation (many of the individuals who hold those values die first).

Economic valuation of nonuse value in monetary terms is limited to stated preference methods based on surveys of what individuals indicate they would pay for a resource in some future context. The most commonly used method is contingent valuation (Bateman et al. 2002, Boyle 2003). Conjoint analysis (Holmes and Adamowicz 2003, Louviere et al. 2000) is much less commonly used for environmental purposes (NRC 2005). Unlike revealed

preference methods, which document actual spending behavior, stated preference techniques depend on personal projections of spending. Both the theory and application of stated preference methods are controversial. Stated preference techniques are thought to produce unreliable results for nonuse valuation by numerous economists (e.g., Hanemann 1994, Portnoy 1994, Boyle 2003), by numerous stakeholders in the outcomes of federal projects, and by at least some decision-makers. Their acceptability has been questioned by government as a consequence (e.g., NOAA Panel on Contingent Valuation 1993). Due to their “conjectural nature” and the difficulty controlling bias using these methods, the Corps has excluded their use for nonuse valuation (USACE 2000).

The concepts of direct and indirect nonuse value are not well-developed (and are not included in Figure 1) but appear to be just as applicable. They might be used, for example, to show a direct and indirect relationship between directly valued endangered species (protected for nonuse value) and indirectly valued ecosystem support systems (protected for the endangered species). Most of the concerned public recognizes value in the species and only indirectly and vaguely in the natural ecosystem context (it typically requires ecological expertise to make the connection). In that respect, the value of the ecosystem support derives entirely from the desire to sustain the species. Consistent with this thinking, the ESA targets species viability as its goal and enlists ecosystem protection and restoration to achieve the goal.

From another perspective, the relationship between ecosystem and individual species is flipped. Ecosystems are viewed by some biodiversity conservancies as reservoirs of both known and unknown biodiversity that deserve protection in their own right to sustain all species, including those that may not be presently known to be at risk (Groves 2003). In that view, the value of individual species flows indirectly from the directly valued ecosystems. The main point, however, is that the total nonuse value is determined by the directly valued resources. To count the sum of both would lead to double counting the value.

Non-instrumental Value

As indicated in Figure 1, some values are excluded from total economic value (in a totally separate box) because they are outside the scope of the total economic value concept, which is based on the tradeoffs that people are willing to make in pursuit of their individual preferences satisfaction

(NRC 2005). Non-instrumental value is not based in utility and cannot be compared directly with or traded for other value. It cannot be measured in units of exchange of any kind. In a project planning context, it cannot be compared with cost to evaluate the net worth of the investment because the value is priceless. Non-instrumental value is often referred to as intrinsic value because it is intrinsic to the valued entity and independent of utility. It is commonly reserved for value held in a human being independent of the value of human labor. Intrinsic value has sometimes been called ethical value because it derives from a moral sense of duty and respect for basic rights, such as the duty to respect personal freedoms and access to opportunities, consistent with certain responsibilities to other human beings. Regardless of status with respect to use and valuation, non-instrumental values are important to recognize because they influence resource use and nonuse, and the values assigned to them.

The concept of human rights is recent in recorded history, and the concepts of species rights and the rights of nature are more recent still (Norton 1986). Recognition of human rights to freedom and opportunities is central to contemporary western law. Many laws encourage respect for and protection of the rights of both present and future generations of people, which underlies contemporary conservation and sustainability philosophy. While there is no explicit right to a clean and otherwise nourishing environment in the US Constitution, several states have adopted such rights (Orr 2003, Lavigne 2003). In one form or another, various state and federal agencies, including the Corps of Engineers, have established an environmental sustainability goal that is based largely on an ethical obligation to future generations of people and on knowing which, if any, of their potential environmental resources can be permanently traded away (totally “consumed” to extinction) to satisfy more immediate demands.

Whereas the intrinsic value revealed in ethical obligations cannot be quantified, the use and nonuse of private and public properties pertaining to those obligations can be negotiated, traded, and quantified in monetary or nonmonetary terms. This includes wild species. In the United States, most mobile wildlife are considered public property and most sedentary life (e.g., plants) are considered private property—except when they are listed for protection under the ESA. At that point, private property rights are transferred to the public trust until such time as those species become secure and are delisted. Evidence of bequest value motivated by an ethical responsibility to future generations is seen in the long range plans of

government and businesses that include maintenance of a heritage well beyond the needs of present generations. With respect to ecological heritage, it is seen in the billions of dollars that have been invested for the protection of biodiversity (Groves 2003) based on priorities independent of present use.

Most people limit their recognition of intrinsic value to human beings, but some also espouse societal recognition of the rights of and implied intrinsic value in other living beings, or even in nature as a whole (Taylor 1986, Noss and Cooperrider 1994, Varner 1999). Some entirely reject quantification of natural ecosystem value (Sagoff 1997). These biocentric intrinsic values are held by some to transcend instrumental value and respect for property ownership (e.g., Bender 2003). However, respect for property ownership is a cornerstone of western law and there is little explicit recognition of the rights of species and nature in western law (Norton 1986). Property rights are explicitly recognized in the United States Constitution (“No person shall be deprived...of life, liberty and property without due process of law; nor shall property be taken for public use, without just compensation”).

Accordingly, conservationists have avoided use of intrinsic value to justify conservation because of its complexity in policy and negotiation settings (Norton 1986; Perlman and Adelson 1997). In fact, biodiversity conservancies rely on respect for property ownership as a primary strategy for their biodiversity protection strategy. Even so, many who back the ESA or invest in species conservation are motivated by a biocentric ethic (Yaffee 1982, Rohlf 1989, Noss and Cooperider 1994).

In the United States, government is expected to work for the general welfare of its citizens—both present and future—as expressed in law, including environmental laws. This may be most clear in Supreme Court decisions pertaining to the ESA, which has been linked more than any other law, at least in its original form, to the biocentric concept of intrinsic value (Yaffee 1982, Callicott 2006). However, Congress later provided for benefit and cost considerations in amendments to the Act, implying that the extinction of a species can be compared to and traded away if the economic losses are great enough to justify it. In working for the general welfare of citizens, the Federal government sets beneficial goals and procedures for achieving them, which usually include some expression of nonmonetary benefit described in the next section.

4 Measuring Non-monetary Value in US Federal Agencies

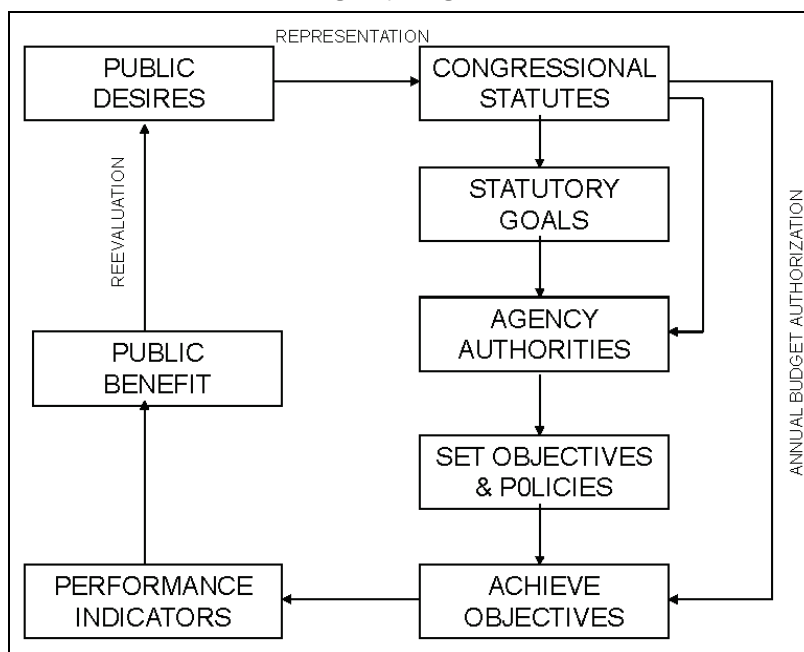
Public Goals Achievement, Cost-Effectiveness, and Public Benefit

Public Goals and Budget Limitations

In federal government, environmental value is usually indicated qualitatively, based on the statutory goals and objectives of laws and executive orders. The public is assumed to benefit when Congress continues to authorize spending to achieve legislated goals and objectives and the administration orders more specific approaches to carrying out those authorities. The goals and objectives identified in laws and orders usually are consistent with prevalent public wishes (Figure 2). With respect to the environment, the goals and objectives usually pertain to protecting and restoring the value of environmental goods and services for present use and/or for national heritage. The public may not benefit if Congress or the President misread or misrepresent public wishes. Repealed laws and amendments indicate a misreading of public desire or a change in public preferences. Laws and orders with long-standing support confirm continued demand for goal achievement. Thus, the best evidence of strongly held desire for public goods and services is found in laws that have been costly but have persisted through time. The intensity of the public desire is also indicated in the annual budget authorization process.

The cost of satisfying the public demand for all public goods and services has in total exceeded public tolerance for increased taxes in recent years. Presidents and many members of Congress have consistently sought to reduce costs by increasing public service efficiency and effectiveness, especially since laws passed during the Johnson Administration significantly increased highly desired and nondiscretionary social spending directed mostly to senior citizens. Coincidentally, Congress entered its most ambitious period of environmental goal-setting in the form of new environmental laws passed during the 1960s and 1970s (Hays 1987, Kline 1997). Growing social obligations to an increasingly larger percentage of the American public enrolled in Social Security, Medicare and other mandatory government spending programs have continued to limit the amount of revenue available for discretionary federal programs.

Figure 2. In the federal government process, public desires are translated into statutory goals. Agency authority establishes objectives and policies that promote goal achievement. Achievement may be measured qualitatively or quantitatively by performance indicators (e.g., acres protected, number of endangered species down-listed). More important is the public sense of improved welfare, which may lead to statute amendment or repeal, or more or less annual funding of the agency program.



The public at this time appears to be more highly committed to maintaining mandatory programs in general than to maintaining discretionary programs. While the goals and objectives of discretionary statutes and tolerated tax load remain about the same, funding to achieve discretionary goals and objectives has decreased from 13.6 % of GDP in 1968 to 7.9% in 2005 in response to public demand for greater social and medical security, the costs of which rose from 3.4% of GDP in 1968 to 11.3% in 2005 (OMB 2007a). Natural resource development funding also has been redistributed to environmental protection and improvement funding at all levels of government. From the tables provided in OMB (2007a), the environmental funding rate has more than doubled the rate of federal tax increases since 1968, while the natural resource funding rate has been more than halved. This redistribution of tax dollars in general appears to reflect the decreasing public demand for the goods and services enhanced by old and new development of public resources and increasing demand for more natural delivery of desired goods and services. Recent interest and growth in restoration of public lands and waters is a consequence of this trend.

The extent to which authorized program funding is actually beneficial is of interest to the Office of Management and Budget (OMB), the agency most directly responsible to the President for managing the executive branch efficiently and effectively. OMB stresses “managing for results” and achieving them through 1) clear definition of program and activity success, 2) clear plans for achieving success, and 3) “a system of accountability” to assure effective program performance as planned (OMB 2007b). OMB policy directs federal program budgeting procedures to consider alternative program plans over the program life cycle and to use benefits and costs estimated in monetary units, when possible, to determine the best plan based on net benefit (OMB 1992).

Cost-Effectiveness

OMB requests that federal agencies perform cost-effectiveness analysis of programs when benefits cannot be measured in monetary terms (OMB 1992), such as for the Corps’ ecosystem restoration program area. When program goals are accepted as beneficial, consistent with legislated goals and authorities, maximum program benefit (achievement of the statutory goal) is obtained mostly by controlling total program costs. The OMB is mostly concerned about program benefit, and from that perspective, cost-effectiveness analysis requires a nonmonetary measure of benefit that can be consistently used across alternative management plans at project and program levels. On an annual basis, when budget funding is almost always limited with respect to program goal achievement, the most cost-effective progress toward program goal achievement results when the selected plan achieves the greatest fraction of the program goal. This cost-effectiveness approach is virtually always elected over a monetary benefits-cost approach for environmental protection and improvement programs and is typically elected for many other programs. Due to of this proclivity, OMB has emphasized intended program results, as indicated in authorized goals and objectives, and management cost control consistent with the Government Performance and Results Act of 1993.

The GPRA authorizes continuous assessment of federal program performance by OMB (OMB 2006a and b). This continuous assessment has been reported to OMB by the federal agencies for several years using its Performance Assessment Rating Tool (PART). Ideally, when general standards are clear, rates of environmental improvement are often reported in various physical measures or percentages of goal achievement that address the general standard. For example, recovery rate of listed

threatened and endangered species to a viable status (and delisting) has been used as a measure of program performance for meeting the goal of the ESA. Environmental improvements of federal lands or waters are often reported in terms of area (acres, sites or fractions of total area targeted); this is common in Department of Agriculture and less so in the Environmental Protection Agency (EPA). Ideally, the increased EQ value per acre should be included, but it remains generally unreported for various reasons. In its annual evaluation of federal program performance, the OMB continues to report that a large majority of environmental protection and restoration programs have not reported metrics or have inadequate metrics (including the Corps). Few, if any, of the metrics are comparable enough to rank the performance of different agencies confidently.

To be consistent with OMB guidance, the best “performance indicators” clearly demonstrate progress in achieving the environmental value as set by statutory goals. Performance indicators of achievement effectiveness typically take the form of specific structural and functional outputs from management. A common approach for environmental gains is some measure of change in geographical area and index to EQ value in that area. Because the quality metrics typically differ from one program to another (if any are reported at all), it is usually impossible to directly compare the efficiencies of most programs. Performance indicators also can be difficult to connect to value when, for example, acres alone are provided without a measure of the quality.

While, in theory, fractions of the environment meeting various standards could be reported, integrating across all standards is problematic. Acres measure environmental area quantitatively, but do not indicate possible wide variation in the quality of each acre. The OMB and the agencies continue to grapple with improving the measurement of program value output through their performance metrics. Numerous publications provide guidance for government performance accounting approaches, some more quantified than others (e.g., Niven 2003), but without widely applicable nonmonetary metrics. Agencies find themselves improvising whatever is appropriate for their needs.

No Common Metric Despite a Common Goal

In one expression or another, either explicitly or implicitly, environmental sustainability is the ultimate aim of virtually all federal land and water management programs. The statutory goals and objectives of the National

Environmental Policy Act (NEPA), the ESA and other environmental laws and directives greatly influenced the evolution and acceptance of this common sustainability goal. The laws express congressional objectives and policies that address aspects and expressions of achieving environmental sustainability (e.g. productive harmony between development and environment in NEPA, species sustainability in the ESA). Because of the commonality, ecosystem sustainability may be one basis for the development of a single metric to compare nonmonetary value realized from ecological protection and improvement across agencies if a widely accepted definition can be settled upon. The challenge is to identify the essentials for achieving sustainability. While it would not be an all-encompassing measure of environmental sustainability, it could contribute significantly.

Motivations for achieving environmental sustainability may not be fully understood, but probably involve a mix of use, nonuse, and intrinsic values. Of the two types of utilitarian value, nonuse values probably are the more important motivation for achieving long-term environmental sustainability, but maintenance of existing use value can also be an important motivation. Indirectly motivating the nonuse value is the obligation to sustain opportunities for future generations. In many different ways, the public has accepted some nonuse of resources to sustain future opportunity through regulation of take and preserves. Many environmental protection laws are motivated in significant part by natural heritage bequest value, option value, and other nonuse values. This motivation is explicit in NEPA and the ESA.

The objectives of most environmental protection and improvement laws are broadly enough stated to provide for use value as well as nonuse value. The Clean Water Act (CWA) for example is framed according to designated use of waters. In that context, the states define the use of waters, ranging from light recreational use (e.g. wilderness use) to intense industrial use, and set standards consistent with use and nonuse needs. At each level of less intense use designation, some use is foregone, and at the lowest intensity of use, nonuse value can be an important motivation for setting aside the waters from any use that might limit future opportunities for use. This includes sustaining the viability of endangered species, which is considered in the designation of water use and associated water quality standards.

This mix of justifying values is nowhere more evident than in laws that authorize habitat protection and improvement such as the Migratory Bird

Act, the Fish and Wildlife Coordination Act, and the Estuary Restoration Act. Most habitat improvement laws were motivated by the desire to protect or enhance use value, such as habitat enhancement for the recreational and commercial benefits associated with certain species. However, nonuse value was typically not precluded from justification of habitat improvement by the requirement for economic benefit and cost analysis. A good example of an exceptional law that explicitly identifies use value as the desired outcome is the Federal Water Project Recreation Act of 1965. It authorizes enhancement of fish and wildlife habitat at federal water resource projects only when monetary benefits of the restoration exceed the costs. Thus habitat improvement authority is rarely a blanket justification for ecosystem restoration for fish and wildlife or other nonuse heritage value. Justification depends on whether the species inhabitants are valued for their present use or their natural heritage or other nonuse value. Many more laws are like the Migratory Bird Act, which provides protections for many common species based largely on recreational use value, but also includes protections for imperiled species with high natural heritage value.

One of the better examples of citizen benefit being linked explicitly to the “the Nation’s heritage” is found in national goals of the ESA. The objective of the law is to sustain species viability and “safeguard” fish and wildlife heritage. The values justifying the law are diverse, but the listing and protection of species under the law sets aside all destructive use based on their heritage value. The utility value justifying protection is indicated in the Supreme Court decision over the snail darter, which determined that Congress was concerned with unknown uses and potential resource value (Callicott 2006).

A large number of physical, chemical and ecological performance indicators have been identified to monitor the environment (Figure 3). The Environmental Protection Agency asked the National Academy of Science to convene a panel to evaluate the scientific merits of ecological indicators (NRC 2000), any of which might be used as performance indicators for achieving national policy goals. The NRC (2000) found that “ecological indicators that describe the state of the nation’s ecosystems and command credibility and attention from the public and decision-makers have been elusive.” Complexity is part of the problem, but the bigger issue has been a lack of criteria for developing national ecological indicators. Most indicators are useful only locally. They examined indicators to 1) the extent and status of the nation’s ecosystems, 2) the nation’s “ecological capital,” and to 3) ecological functioning or “performance.”

Figure 3. Example performance indicators for environmental monitoring.

Recommended Indicators (NRC 2000)	
Extent and Status Indicators	
	Land cover
	Land use
Ecological Capital	
	Total species diversity
	Native species diversity
	Nutrient runoff
	Soil organic matter
Ecological Functioning	
	Carbon storage
	Production capacity
	Net primary production
	Lake trophic status
	Stream oxygen
	Nutrient-use efficiency
	Nutrient balance

The charge of the NRC panel did not include differentiating indicators of value that could be expressed monetarily from other values. Most of the NRC indicators simply indicate ecological change without clear connection to value. Land cover and use alone, for example, indicate value grossly at best. That is also true for indicators of ecological functioning. They all depend on more detailed specification of use and non-use. Ecological capital is intended to indicate the provision of goods and services. Species and soil condition “drive and maintain ecosystem processes.”

But these goods and services indicators are not separable into use and non-use value without more specific information, and are of little use for indicating nonuse objective achievement. Similar limitations are found in the indicators provided by The H. John Heinz III Center for Science (2008). These reports are useful for identifying indicators having widely

available data. Any of these indicators might be considered for use in specific agency settings to indicate achievement of specific statutory goals and agency objectives such as setting standards for water quality in the Clean Water Act (nutrient runoff) or native species diversity in the ESA. They are not readily compared, however, and do not facilitate integration of efforts to protect and restore environments of greatest value.

Environmental authorities are typically programmatic. The project-based authorities of the federal water resources development agencies are key exceptions. The differences are significant and can affect interactions among the agencies, which are often collaborative and sometimes antagonistic. Some difficulties can emerge over apparent differences in concepts of value; this often revolves around whether or not value can or should be measured in monetary terms.

More specific differences in environmental program goals and objectives fall into two categories: regulatory programs and resource management programs. These are described below with respect to how benefits are measured.

Programmatic Authorities

Regulatory Programs

Few environmental regulation laws require a benefit-cost analysis. Rather, they are usually directed at effective technological solutions to conditions that are out of compliance with pollution regulations and environmental standards (Portnoy 1990). The environmental regulation functions of federal government are carried out by the Environmental Protection Agency (EPA) with important supporting additional roles played by the Fish and Wildlife Service (FWS) and National Marine Fisheries Service (NMFS) in the federal departments of Interior and Commerce, and by the Corps of Engineers. The Council of Environmental Quality (CEQ) coordinates federal environmental programs and new policy development, reports annually to the President on the condition of the environment, oversees the execution of NEPA, and resolves interagency disputes over the NEPA process. NEPA has been a unifying force in the way federally funded organizations approach project and program planning to sustain environmental values, primarily through management of cumulative environmental impacts (e.g., Caldwell 1998).

Whereas NEPA establishes both protection and restoration of EQ in its congressional goals, the NEPA process as practiced has focused on sustaining existing environmental value; i.e., “to promote efforts which will prevent or eliminate damage to the environment and the biosphere” (Caldwell 1998). In this regard, NEPA does not distinguish value that can be monetized from value that cannot. Consequently, the quality of the environment for outdoor recreational use is included along with natural heritage and other use and nonuse value of significance. Analyses required by NEPA in the development of an Environmental Impact Statement (EIS) must include examination of alternative plans for their beneficial and adverse EQ and the estimated costs of avoiding, minimizing and compensating for adverse impacts.

No specific standards are to be met under NEPA. What is actually done to protect the environment results from agency sense of responsibility to the general standards established by NEPA goals and to the pressure of other agencies and the public during the review process. That pressure is the key to any success claimed for goal achievement. The EPA manages EIS reviews by relevant agencies and comments on completeness, accuracy, and other aspects related to the goals and process, and to their relationship to the objectives of other environmental legislation. At the heart of the review process is the extent to which environmental values are sustained through impact mitigation.

Central to the concept of impact on EQ is the significance of any effect on environmental resources. Important determinants of significance are the scarcity and distinctiveness of the environmental resources impacted and the reversibility of the impacts. These often are resources that are protected by laws other than NEPA. However, regardless of its legal status, strict interpretation of NEPA policy encourages decisions that would preserve any important cultural and natural resources threatened with total loss. One means for identifying resource importance is in policies set forth in law. For example, the ESA establishes that all species are important enough to maintain their existence in fish and wildlife heritage. Environmental policy in NEPA encourages avoidance of further damage to those vulnerable species even if they are not officially listed as threatened and endangered. Only a small fraction of those determined by conservation science to be vulnerable to extinction are listed under ESA protection (Scott et al. 2006).

In general, impact avoidance is urged when distinctive resources grow more scarce and insecure from total loss, and additional negative impact on them is very costly or impossible to reverse. Impact avoidance offsets resource use to protect the resource's nonuse value, which in NEPA is the value that justifies preservation of natural and cultural aspects of national heritage. Reversible impacts on more common resources may not require total avoidance as long as the local losses in value can be confidently replaced through compensatory action, including restoration of damaged sites and creation of new sites. These common resources are usually valued for present use (e.g., recreation), and they are relatively easy to replace with substitute resources nearby.

The uncertainty associated with assessments of impact intensity, extent, and reversibility is an important aspect of the NEPA process. The risk of permanent loss of valued resources is supposed to be managed by careful analysis of cumulative effects and their mitigation. This uncertainty is a critical issue in environmental impact analysis. Environmental advocates are willing to accept the risk that protection costs are too high for optimum benefit to the nation. Development advocates seeking more immediate benefit are more willing to accept the risk that irreplaceable and distinctive resources of potential high value will be lost. This tradeoff between present and future benefits and costs is central to most reviews even if it is not explicitly expressed as such.

NEPA by itself establishes no standards by which to determine when effects are not significant or must be avoided, minimized and compensated. Other laws, passed mostly after NEPA, have made up for that deficiency by establishing general and specific standards based on the assumption that the public is benefited by their achievement. The Clean Water Act is a good example of legislation that sets general and more specific water quality standards, which serve the policies and goals of NEPA. Water quality standards set by the states and the EPA establish state objectives for achievement. Different standards are set according to the designated use of waters. This minimizes the opportunity costs incurred by establishing universal standards that are not universally beneficial. Each state develops a plan for achieving objectives cost effectively. The tradeoffs between small negative effects and high avoidance costs are considered during standards and plan reviews by state and other government agencies, and the public.

Once standards are established, the general assumption is that meeting them will add value beyond the costs of implementation. As such, meeting them in the most cost-effective manner is most beneficial. The extent to which the general standard is achieved at the program level depends on the achievement of each water quality standard and can, in theory, be relatively easy to report in acres, fraction of total goal achievement or (less satisfactorily) number of sites. So far in OMB performance reporting, the EPA programs related to water quality improvement under the Clean Water Act report the number of sites partially to fully improved. However, the connections of this physical measure to use value and nonuse value are not distinguished and are not clear. This is a common problem encountered in environmental performance reporting to OMB.

A number of biological metrics have been developed with the intent of measuring the “integrity” of aquatic ecosystems (e.g., Hilsenhoff 1987, Plafkin et al. 1989, Karr and Chu 1999) in response to the primary objective of the Clean Water Act: to restore the chemical, physical and biological integrity of the nation’s waters. Most of these integrity indices are based on the assumption that biotic communities are the most sensitive indicators of environmental condition. Such indicators develop a relationship between the composition of biotic communities and the condition of the inhabited environment along a gradient of human-caused variation from the natural condition. They indicate resource change but vary from region to region and need to be regionally developed (e.g., Barbour et al. 1996). Like all other environmental indicators, they do not indicate value until they are related to goals and standards established by law or other official action. Little attempt has been made to quantify the relationship between integrity indices and use or nonuse value at the national scale because the indicators vary regionally. They do not seem to have been used to report agency annual performance to OMB.

The CWA prohibits discharge of soil, sand and other materials into the Nation’s waters without a permit. The Corps and approved state programs are authorized to deny or issue permits, and to attach conditions, such as compensatory mitigation, for significant environmental damage. As it relates to the objectives of the CWA, the benefit from this program depends on the extent to which protected and replaced wetlands function to maintain water quality standards as well as to maintain other human services (e.g., flood damage reduction, habitat provision for desired species). The Corps expects permit applicants to avoid discharge into

waters where feasible, to minimize discharge that is unavoidable, and to provide compensatory mitigation for significant loss when it occurs, as determined with respect to the objectives of the CWA. Avoidance is typically dictated by compliance with existing law, such as the ESA. In that respect, the presence of an endangered species determines that the nonuse value is too high to permit discharge use. The Corps explicitly reports the number of acres in which discharge alteration was avoided, the number of acres altered, and the number of acres for which alteration compensation was paid. This satisfies OMB reporting needs, but does not indicate the value of each acre for either resource use or nonuse.

The environmental value preserved in compensatory mitigation has been indicated in a variety of nonmonetary ways that are conceptually quite similar. The basic concept requires an indicator of environmental value that is used first to characterize the environment before and after it is degraded, and then used to guide creation or restoration of a replacement environment commensurate in value. Even though the environmental value may be measurable in monetary units, monetary value is rarely estimated. The emphasis is placed instead on replacing the lost value based on some indicator that can be used to guide the replacement. Two types of metrics in particular have been used to guide permit applicants toward successful mitigation: the Habitat Evaluation Procedure (HEP) and the Hydrogeomorphic Method (HGM).

The HEP and, to a lesser extent, the conceptually similar Instream Flow Incremental Method (IFIM), were developed under the lead of the US Fish and Wildlife Service (FWS 1980, 1981; Bovee 1982) to facilitate compensation for habitat loss from development. The HEP has been modified for specific state and other use. It and IFIM were used to carry out mitigation provisions of the Fish and Wildlife Coordination Act as well as to guide mitigation under Section 404 of the CWA. The HEP integrates a measure of habitat quality indicated by a habitat suitability index (HSI) with a measure of habitat geographical area (acres) to create a habitat unit (HU) indicator of relative habitat value. For either HEP or IFIM the habitat value has no meaning outside the context of the destroyed habitat and the habitat that is created or restored in compensation. The HEP, in particular, has been adopted for ecosystem restoration planning. While it has been used for comparing plans within projects, there is no rational way to compare values across different projects.

The HGM (Smith et al. 1995) was developed specifically to aid the Corps' execution of its regulatory program for Section 404 of the CWA. The method includes establishing a wetland classification based on wetland functions and establishing standards based on the functional capacity of natural and modified wetlands. A functional capacity index and acres impacted are integrated to characterize the lost value and compensation value for wetlands damaged by projects much as the HU is used to characterize the relative value of habitat. A variety of index approaches have been developed based on the same fundamental concept (e.g., O'Neil et al. 2001, Stakhiv et al. 2003). All of them, including the HGM, have the same comparability limitations as the HEP. They do not allow OMB or any other party to compare the value added by different agencies or to sum the value added for all federal agencies.

The ESA is one of the few environmental laws or executive directives that was clearly intended for sustaining nonuse value as expressed in an intact fish and wildlife heritage. It establishes a broad standard in the national desire to achieve plant and animal species viability in all but a few pest insect species. It authorizes the establishment of more specific standards for the protection and recovery objectives, including declaration of critical habitat, for each species listed under the Act. The law directs that a species be listed based only on their vulnerability to extinction (see Rohlf 1989 for a detailed early legal history of the law) and independent of use value. This direction clearly serves nonuse value by setting aside the species from jeopardizing use (including use of critical habitat), based on no other criteria than the threat of extinction; i.e., their viability is not secure. In effect, the protection offered is of the "last resort" because the Act provides no means for preventing the need for listing in the first place or relisting once a species has recovered and been delisted (NRC 1995).

The OMB assesses the value of federal regulatory programs by requiring a benefits assessment of them (OMB 2003) and by requiring annual performance assessment reports. The purposes are to assure that the costs incurred by regulatory programs are justified by the benefits produced and to encourage a search for the approach that generates the most benefit to society. The primary tool advocated for regulatory analysis is benefit-cost analysis; in this model, all benefits and costs are ideally quantified in monetary units. However, the OMB recognizes that it is rarely possible to express important regulatory outcomes, including environmental protection and improvement, in monetary terms (OMB 2003). In those and similar

circumstances, it directs the quantification of outputs in nonmonetary terms and recommends that evaluators “exercise professional judgment” to determine how important the benefits and costs are in the context of benefit-cost analysis. The analysis recommended by OMB also includes an estimate of the minimum nonmonetary benefit required to justify all costs incurred by the program (OMB 2003). One shortcoming with the annual program performance reporting to OMB is that mitigation actions are often included among the costs of development programs but are not separately reported in confirmed environmental units of damage and compensation. Consequently, there is no way to determine whether all costs are fully counted.

Independent reviews (Portnoy 1990) in general confirm that the costs of environmental regulation are high and the benefits are difficult to completely assess in monetary terms. Rather than second-guessing the value of most standards, Portnoy emphasizes ways to be more cost-effective. He believed it is not in the interest of Congress to use a benefit and cost balancing approach because of the complexity implied and the fact that economists have not done a good job making their case for such an approach. It is as if Congress assumes that a benefit and cost balancing approach implicitly occurs in the federal budget process as programs compete for federal funding—i.e., the programs having the greatest public support (because of the perceived benefits) are more likely to be funded at a certain request level or higher. OMB has had limited success developing benefit-cost guidance that gets implemented for environmental programs. The criteria used for tradeoff analysis across programs remain unclear.

Resource Management Programs

Most federal agencies derive their authority for improving EQ through the natural resources they hold in trust, operate and maintain for public use and benefit. Over one third of the geographical area of the United States has been managed by federal agencies -- in principle for sustainable use of renewable resources -- since the early 20th century (Hays 1957). The concept of sustainable use was based largely on general knowledge of the resource stocks required to sustain production at some desired level for use primarily within a human generation (about the next 25 to 35 years). Sustained yield management of public resources was motivated by rampant exploitation of timber, recreational wildlife, livestock forage, and water resources during the late 19th century. It centered on regulating harvest to assure resource sustainability for specific uses. It integrated ecological and economic

concepts to estimate specific resource supply and demand. It had more in common with option value that can be measured in monetary terms than in cross-generational bequest value. But setting aside a natural resource heritage to sustain opportunities for future generations was an additional motivation held by a smaller but influential fraction of the public, including Theodore Roosevelt and other leaders instrumental in establishing the basis of existing federal land and water management policy (Hays 1957).

Heritage value has since gained in importance with regard to natural resource management. It has become an increasingly important justification for establishing the National Park System of the United States, and for protecting and restoring EQ. Heritage is an explicit justification of investments in the EQ considered under NEPA and the species viability standard of the ESA. In those contexts, resource security from permanent loss and resource distinctiveness are important criteria for determining resource scarcity and national heritage preservation needs. Most federal land management agencies have set aside some authorized use of public resources (e.g., timber, grazing, intense recreation) to protect and recover EQ. In policy, and increasingly in practice, the land management agencies have largely moved beyond the concept of sustained use of ecological resources (e.g., forest, range, wildlife, fisheries) to ecosystem-based sustainability of all ecological resources. This transformation reflects growing interest in sustaining opportunities for future generations (much encouraged by NEPA). By focusing on the heritage of future generations, options are also sustained for the present generation.

In large part because the effects of federal land management on the environment manifest themselves in water quality and biodiversity changes, the CWA and ESA have substantially influenced how agencies plan to protect and restore EQ consistent with the NEPA process. With respect to water quality, most have adopted a watershed-based planning approach with some reaching back to foundations established more than a century ago (Cole et al. 2005). The watershed approach is also a strategy for restoring habitat of some aquatic species (e.g., NRC 1996). But decreasing discretionary funds in the federal budget have led to limited investment in the recovery of ESA-listed species, less attention paid to reversing species declines before they need to be listed under ESA protection, and fewer discretionary heritage preservation investments by federal agencies.

Emphasis on cost control leaves little for management discretion other than what is obligated under the law and through political pressure. For example, less has been done than might be by federal agencies to contribute to the recovery of listed species in large part because the ESA neither obligates contributions (Rohlf 1989) nor spells out clear standards to be met (Suckling and Taylor 2006). Indeed, the listing of species has waxed and waned depending in part on cost consciousness and is far fewer than the number of species actually vulnerable to extinction (Scott et al 2006). The pressure to do no more than what is necessary to meet ESA and other requirements of environmental law has resulted in numerous lawsuits brought against federal agencies by NGOs to assure that at least the minimum is provided. Thus, much of what has been done to manage for EQ is determined by cost avoidance, including avoidance of lawsuit. In indirect fashion, this fits with the policy of OMB (1992), which typically emphasizes the least cost approach to meeting the minimum required by Congress, except when the President directs otherwise.

Attention paid to especially significant environmental impacts sometimes does generate extra funding as long as the costs are not exceedingly high. For example, change in the operation of Glen Canyon Dam by the Bureau of Reclamation exemplifies a NEPA- and ESA-inspired change in a federal property management plan (NRC 1999b). The dam acts as a sediment trap for a river ecosystem that evolved in the presence of large amounts of sediment shaped by flood events into a diverse topography of bars, beaches, and islands. Without flood events, the sediment gradually redistributes into a much flatter bottom form below the river surface, which degrades recreational value and habitat for imperiled species. In 1996, a flood pulse was created by releasing more water from the dam. It had a short-term effect on sediment topography and uncertain effect on recreational and habitat value (NRC 1999b). Additional flood events were considered, but low water storage drove up the cost of using water for flood pulses and delayed subsequent flood-water releases.

Little attempt had been made to assess benefits of federal programs using a quantitative measure of the value gained until OMB began requiring program performance assessment reports under the authority of the Government Performance and Results Act (GPRA). Until that point, the achievement of goals and objectives established in law were simply assumed. In natural resource agencies, priorities for achieving goals are typically determined in management plan objectives. These objectives are

often established at the local management-unit level based on inventories of need. They were typically funded from the program operation and maintenance budgets and based on requests from the local level. They were expected to reflect the least-cost requirements for achieving objectives as directed by OMB (1992). Most agencies have used physical measures of output (such as the number of acres affected) as program performance indicators, but they are typically difficult to link to use or nonuse value added, since they rarely discriminate between the two options. While the added expense of devising more informative indicators may be an impediment to developing them at this time, there may be more fundamental deficiencies in our understanding of the relationships among management plans, plan implementation, resource output and national benefit.

The priority-setting process in federal land management plans is not based on common standards despite recommendations by a National Research Council committee as requested by Congress (NRC 1993). The Committee confirmed that most successful acquisition criteria have well-understood policy goals and advised that these goals be organized into national ranking systems. It left identification of ranking criteria to the agencies, but advised plan coordination and careful analysis of resource protection gaps to determine where the greatest land acquisition needs were to achieve policy goals. The OMB devised an approach to land acquisition for conservation purposes based on points assigned to categories that reflected administration priorities, which often change with administrations (anything but standard). Consistent with OMB history, it also included a cost minimization category (NRC 1993).

Interagency management of coastal area resources also focuses on general and specific standards, and plans for meeting them, without separately quantifying use and nonuse value added. For example, the Estuary Restoration Act authorized the establishment of an Estuary Habitat Restoration Council charged with developing a strategy to “maximize benefits derived from estuary habitat restoration projects” (Sec. 106 (a)). The Council established a general standard in a goal to attain “a self-sustaining system integrated into the surrounding landscape” (Sec. 103 (4)). In pursuit of this standard, the Council supports projects “developed in an ecosystem context with multiple benefits and that utilize natural processes to restore and maintain estuarine habitat” (Federal Register 2002). Therefore, whether or not the benefits are realized is indicated by

meeting the “self-sustaining system” standard. To be consistent with OMB (1992) guidance, the least cost approach to realizing the final goal would be the most beneficial approach to meeting the standard. The value justifying restoration investments includes both use value (commercial and sportfishing) and nonuse value (recovery of imperiled species). However, success is indicated by a self-sustaining system standard for restored ecosystem; not by indication of use and nonuse value. For various reasons, the standard may or may not indicate improved use value or improved maintenance of natural heritage.

An example of more complex law that provides for a mix of use and nonuse values in a land and water management plan is the North American Wetland Conservation Act (NAWCA). The Act facilitates achievement of objectives identified in the North American Waterfowl Management Plan (NAWMP). The major purpose of the Plan is to enhance recreational value of hunted and watched waterfowl, and other wildlife by increasing and stabilizing their abundance. The NAWCA serves the objectives of the Plan by authorizing protection and development of waterfowl habitat that meets certain standards. It is also concerned with restoring habitats that support bird species in initial decline toward levels of abundance that require listing under the ESA. The NAWCA authorizes redirection of tax revenue on recreational equipment to wetland habitat creation, rehabilitation and restoration consistent with the NAWMP. Benefits accrue with achievement of Plan recreational objectives. Achievement rate depends on the rate of recreational equipment purchase. Progress is indicated by acres of wetland conserved and wildlife use rates, which do not necessarily equate with increased recreational use benefit or heritage maintenance.

In summary, the programs of all federal agencies are encouraged by NEPA to protect use value and nonuse value associated with maintaining natural heritage for future generations. Some agencies are authorized to improve both use value and nonuse value. Agencies are not required to differentiate or separately measure use and nonuse value in program performance measurement. Typically, agencies use some type of resource condition standard to gauge performance of programs under the assumption that meeting that standard will be nationally beneficial. The standards are typically based on what the agency determines to be a publicly desired resource condition. The least cost approach to achieving the standard is favored in principle. Measurement of monetary benefits derived from resource use or resource heritage value is rarely attempted. Program

comparison for efficiency and effectiveness is complicated by the number of diverse and inconsistent program performance standards and measurements of success.

Project Authorities

Environmental Protection

Project-authorized development by the federal water resources development agencies is an exception to the more general rule that federal management of natural resources and the environment is funded at the program level. With the possible exception of the Tennessee Valley Authority, Congress never established comprehensive program goals in organic legislation for the federal water resource development agencies. In addition to TVA, these agencies include the Corps, the Bureau of Reclamation, and the Natural Resources Conservation Service. Agency purposes were authorized (navigation, irrigation, hydropower, flood control) to serve national economic development goals that were to be pursued through individually authorized projects. More recently, Congress authorized small continuing authority programs for the primary purposes of Corps water resources management (navigation, flood damage reduction and ecosystem restoration), allowing some small project funding discretion within agencies. Even in those programs, members of Congress often direct project funding through legislation earmarks.

Also exceptional among Congressional requirements of agencies is the legislated emphasis on quantitative measures of costs and benefits from federal water resources projects. While benefit assessments have been made on an ad hoc basis since the 19th century (Shabman 1997), Congress required water resources agencies to assure that benefits exceed costs in the Flood Control Act of 1936 (Feldman 1991, Shabman 1997). Particularly close attention has been paid to justification of federal water resource agency budgets since the 1930s, when a much larger fraction of the Federal budget went to water resources development.

Under the Water Resources Planning Act of 1965, Congress authorized the formation of a Water Resources Council (WRC) to develop guidance for federal water resources planning and objective achievement at the local project, regional, and national levels. Congress also identified four overlapping national objectives, but only project-level planning guidance was actually developed (WRC 1973, 1983). The guidance directed

measurement of all change in resource value that flowed from project effects in monetary terms when possible and in nonmonetary terms otherwise. Since it is resource-based, the value flow considered in objective achievement is limited to utility value, which can be compared across objectives and traded in pursuit of maximizing project benefits. That utility value includes both use value and nonuse (or existence) value.

Guidance for the Act identified a six-step process: 1) identify problems and opportunities, 2) inventory and forecast conditions, 3) formulate alternative plans, 4) evaluate alternative plans for their justifiability, 5) compare alternative plans for relative benefit, and 5) recommend the most beneficial acceptable plan (WRC 1983). Under that federal guidance, planning incorporated the NEPA process, which focused on considering the need to mitigate significant adverse effects through avoidance and minimization of effect and by compensation for any remaining adverse effect. The guidance was completed before any federal water resources agency was authorized to improve EQ and did not provide guidance for that event. The Corps developed its own guidance in 2000 consistent with the goals and objectives of the WRPA and its new EQ improvement authority.

The federal planning guidance for the WRPA defined EQ to be considered under NEPA as natural and cultural resources with ecological, esthetic and cultural attributes. Health and safety effects are considered separately, but are not included under EQ. Most relevant for EQ improvement authorities that came later to the Corps, was the definition of the ecological attributes of natural resources that qualify as EQ:

“Ecological attributes are components of the environment and the interactions among all its living (including people) and nonliving components that directly or indirectly sustain dynamic, diverse, viable ecosystems. In this category are functional and structural aspects that require special consideration because of their unusual characteristics.” (WRC 1983)

From the standpoint of EQ protection, the key concern in the planning guidance is that the ecological attributes contribute to sustaining “diverse” and “viable” ecosystems. The key to discriminating the relative importance of a particular ecosystem’s attributes is discerning how unusual the characteristics are. The emphasis on unusual resource attributes implies the decision-making importance of distinctive, rare, and irreplaceable traits that are important among NEPA criteria for determining significant EQ. It also indicates the basis of resource value and its scarcity with respect to public demand. When that scarcity is based in a demand for use,

it usually can be measured in monetary terms. When it is based on a demand for setting aside a resource from present use, it is much more controversially measured in monetary terms and many do not accept the validity of the results (NRC 2005).

The federal guidance for project planning (WRC 1973, 1983) directed the agencies to measure any of the natural and cultural resource value that could be measured in monetary terms to be so measured, compensated, and included among project costs following the NEPA process (Many ecological attributes may support resource value that can be measured in economic terms such as commercial fisheries or recreational wildlife). High costs may exclude projects from further consideration on economic grounds. Numerous methods for direct and indirect monetization of use value are accepted (WRC 1983). Corps project planning policy prohibits nonuse values to be measured in monetary terms using the one non-experimental method available (USACE 2000) because the results are too uncertain.

Under federal guidance for project planning, the remaining natural and cultural resource value that cannot be acceptably measured in monetary terms (mostly, if not entirely, nonuse value), must be measured in some quantifiable nonmonetary terms, evaluated for environmental significance and sustained if that significance warrants it (WRC 1983). The costs of sustaining the value are borne by the project and may eliminate a project from further consideration based on the costs. The guidance only addresses the instrumental value that can be compared and traded off in a project planning process that seeks the most beneficial plan. Like other decision processes in government, it does not address intrinsic values as objectives because they are nonnegotiable. To the extent they show up in restrictive laws, they act as planning constraints.

The fundamentally different qualities of ecological, cultural, and aesthetic attributes indicated improbable development of an acceptable single metric for all nonuse value captured in EQ. However, incommensurate value was not much of an issue before the Corps gained EQ restoration authority, because protecting significant resource value could be done through physical avoidance, minimization and compensation without estimating the benefit explicitly. It only had to be “significant” enough to warrant the value protection. To this day, there is no consolidated measure of the utility value sustained under NEPA and other environmental law, nor is there likely to be one in the foreseeable future because of the complexity of measurement

and inadequacy of information. It may be possible, however, to develop a single ecological metric for EQ improvement using an ecosystem restoration and protection approach as authorized for the Corps.

EQ Improvement Using an Ecosystem Restoration Approach

No other agency has an authority quite like the EQ improvement authority of the Corps of Engineers. Understanding that authority and the policy objective statements that follow is essential for choosing or developing a meaningful benefit metric for project planning and for setting annual priorities for project funding. The complexity of ecosystem restoration policy evolution may have contributed to past difficulty in finding suitable non-monetary metrics for indicating environmental benefits. The ecosystem improvement authority was passed in the 1996 WRDA (Section 206) and is grounded in the EQ protection and improvement objective of the Water Resources Planning Act of 1965. The 1996 legislation programmatically authorizes the Corps to carry out aquatic ecosystem restoration and protection to cost-effectively improve EQ in the public interest.

The definition of EQ had its genesis in the interpretation of the WRPA's EQ protection and improvement objective into federal guidance (WRC 1973). At that time, it was defined in terms of environmental enjoyment and national heritage protection and enhancement. Due to the fact that it was restricted to an ecosystem restoration and protection approach, the EQ improvement authority of 1996 pertained only to ecological aspects of natural heritage. The Corps does not permit measurement of nonuse value, including heritage value, in monetary terms using contingent valuation (USACE 2000), the only existing non-experimental technique. Therefore, the heritage value justifying ecosystem restoration investment must be measured in nonmonetary terms.

The protection aspect of the required ecosystem restoration and protection approach to authorized EQ improvement indicates the importance of long-term sustainability for heritage maintenance. Reflecting this interpretation, Corps planning guidance (USACE 2000, pg 2-1) explicitly defines environmental protection in terms of heritage preservation:

“Protection of the Nation’s environment is achieved when damage to the environment is eliminated or avoided and important cultural and natural aspects of our nation’s heritage are preserved.”

For the Corps, the authorized means of eliminating damage is ecosystem restoration. The heritage preservation wording is also found in the policy and goal statement of NEPA.

Most other government programs provide little insight into how the Corps should measure the value of an improved natural heritage condition that results from ecosystem restoration because they do not discriminate between use and nonuse values. This stems largely from the typical mix of use and nonuse values embedded in the national objectives of federal environmental law (including NEPA and most fish and wildlife law). The Corps is uniquely required to justify restoration investments based on values that cannot be acceptably measured in monetary terms. This exceptional circumstance was created out of the prerequisites that emerged from the Water Resources Planning Act and other laws; the unique EQ restoration authority granted to the Corps also played a role in shaping this requirement. It is the basis of much confusion among agencies and within the Corps when these personnel try to identify the precise national objective of the Corps' restoration projects.

Due to its unique authority, Corps policy guidance for EQ improvement under its ecosystem restoration mission provides the most useful indicators of what is nationally significant among the many useful resources produced by ecosystems. The Corps incorporated an ecosystem restoration mission into its project planning regulations (USACE 2000) a year after it created an "ecosystem restoration purpose" and associated regulations (USACE 1999). Those regulations explicitly exclude aesthetic and cultural resources and identified a more naturalistic condition of significant ecological resources as the target of EQ improvement. The planning guidance (USACE 2000) followed suit. The substance of the ecosystem restoration purpose statement was incorporated into the first sentence of the Corps' study objective statement. While the statement linked the improved state of resource degradation to restoration of a more natural condition, the next implied that a more naturalistic condition of the ecosystem is permissible as long as it reasonably mimics the conditions that would have naturally occurred.

"The objective of ecosystem restoration is to restore degraded ecosystem structure, function, and dynamic process to a less degraded 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." (USACE 2000)

This point is important because it indicates that the targeted ecological resources (in the form of structure, function and dynamic process) do not have to be restored to a more natural quantity and/or quality by removing human effect. It allows the simulation of the supporting ecosystem functions and structures, such as modifying flows below a dam or building a fish ladder over it, to be more like the natural ecosystem condition, but not necessarily more natural. While simulation adds human effect instead of removing it, a simulated natural condition can result in desired resource outputs that are more like a condition without human effect, consistent with the study objective. It is also clear in planning guidance (USACE 2000) that full restoration is not required (in fact, many ecologists would say this is impossible due to the pervasive and virtually permanent impacts of humanity).

The objective statement of the planning policy guidance then provides more insights into which ecological resources are desired resources:

“Indicators of success would include the presence of a large variety of native plants and animals, the ability of the area to sustain larger numbers of certain indicator species or more biologically desirable species, and the ability of the restored area to continue to function and produce the desired outputs with a minimum of continuing human intervention.” (USACE 2000).

These “indicators of success” are the only benchmarks provided in this planning policy guidance. The indicators of success are the indicators of value added that justify the ecosystem restoration and are critical to understanding what a quantitative metric should indicate. Without the indicators of success to determine what is specifically valued in restoration of a more natural condition, virtually any return to a condition mimicking less human effect might be construed as justifiable. That would non-sensically imply that all water resources development, if it ever was beneficial, is no longer beneficial enough to exceed the cost incurred by creating a less natural condition. To make more sense, reducing ecosystem degradation has to be quantified in terms of the ecological resource value to be gained by restoring a condition to a more natural ecosystem condition, not by the more naturalistic condition itself.

Corps planning guidance relies on the concept of resource significance to sort out justifiable investments. It was first used in federal project planning guidance (WRC 1983), but Corps planning guidance (USACE 2000) never mentions the word “significant” in either the federal project planning objective or the ecosystem restoration study objective. It instead

refers to desired ecosystem resources. The ecosystem resources are desired in greater quality or quantity by the public as indicated in the objectives of law and other evidence.

The information that explains what is degraded—what is *desired* by the public to replace lost value—is in the description of “indicators of success.” Only one indicator of success clearly indicates a response to public *desire* for more of something, i.e., *more* biologically *desirable* species. Determining what that emphasis means is critical to determining an appropriate metric for sustaining natural heritage value—the EQ value that is the unidentified, but apparent Corps objective for ecosystem restoration.

In contrast with economically desirable species that are quite evidently desired for their use, biologically desirable species could be interpreted to mean species desired by people for their intrinsic value (a biocentric point of view). But a focus on improved intrinsic value is inconsistent with the Corps’ policy focus on utility value. On the other hand, improvement of natural heritage value, a nonuse utility value, can be and is routinely traded off for improvement of resource use value. Heritage maintenance is the usual institutional intent of setting aside destructive use of resources, as in NEPA and the ESA. The heritage value of natural resources has little value when all of the resources are abundant and replaceable (and can be regenerated without fear of total loss). It gains value as resources approach total loss and become irreplaceable. Its value depends on its scarcity with respect to some level desired by the public, such as the species sustainability criterion established in the ESA.

In contrast with restoration investment, protection law intends to stop tradeoffs of highly valued natural heritage and becomes a constraint on project objective achievement. Investments in restoring unsustainable natural heritage to a secure status are tradable, but there is no bright line for determining when heritage value trumps use value. The decision is subjective and history indicates that use value tends to be favored over nonuse value. Many species are in decline toward extinction as a consequence (Cole 2009). Most Corps restoration projects so far have occurred where use value is low or where it is high but compatible with heritage maintenance (e.g. light recreational/tourism use). This tendency can be seen in the outcomes of the ESA. Protection of threatened and endangered species has tenuously kept many listed species from

extinction, but restoration of those species to a status secure enough to delist them has been much less successful (Scott et al. 2006, Cole 2009).

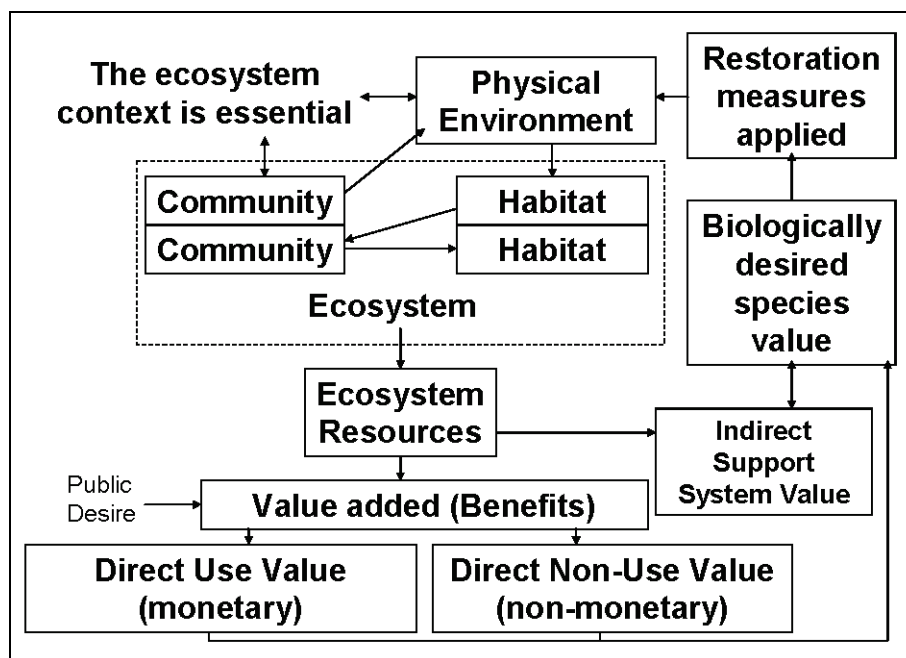
The attention paid to the “large” variety of native plants and animals in the study objective statement indicates that the biologically desirable species and their support communities are or should be native species and that their restoration should at least not reduce the native diversity. This could be the key to understanding the importance of biologically desirable species. Sustaining global biodiversity is the underlying motivation for emphasizing native species and it is generally believed that this goal is most likely to be achieved by restoring native biodiversity to a sustainable condition in more or less natural ecosystem settings. In that light, biologically desirable species are selected for their biological distinctiveness independent of any use value, because they contribute to the maintenance of the natural heritage of naturally diverse ecosystems. The study objective indicates that the “desired ecosystem resources” are species desired for their unique biological attributes – attributes which contribute to native species diversity. These are the same nonmonetary criteria used to list species under the protection of the ESA.

The other two “indicators of success” are not indicators of desired output. Indicator species indirectly indicate the value of the desired outputs, but have no independent value of their own. The ability of a region to support the desired outputs also has no value independent of the desired outputs. The study objective statement leaves the door open to whether or not other outputs might indicate success. Because no other possibilities are described, but are not precluded, the objective remains incompletely determined. Whether or not other possibilities actually exist, however, it is difficult to conclude that the Corps’ ecosystem restoration objective is not largely about restoring biologically desirable species (desired for their biological as opposed to economic attributes) through restoration of degraded ecosystem support based on the nonuse value of the species living naturally in their ecosystem context. A nonmonetary metric based largely on this nonuse concept is likely to capture much, if not all, of the ecosystem value.

Consistent with this interpretation of the ecosystem restoration objective, restoration measures are organized into plans formulated to reestablish native, biologically desirable species through reestablishment of more natural hydrology and channel and basin morphology in the proper ecological context. Placing the restoration in a proper ecosystem context is

essential if the desired species and naturally supportive ecosystems are to colonize and persist in the restored project area (Figure 4). The result produces both biotic and abiotic resources most likely having both monetary and non-monetary value. The plan evaluation looks for a significant reestablishment of the biologically desirable species, which are the directly valued and necessary outputs from restoration projects, if the projects are to qualify for project implementation funding. The value of the supporting ecosystem is indirectly determined by the value of the desired biological species and is not counted separately. Simply restoring a more natural “supporting condition” without recovery of the biologically desirable species has no value that counts toward the ecosystem restoration objective.

Figure 4. This schematic for the ecosystem restoration concept of the Corps illustrates some of the primary considerations. Based on its authorities, ecosystem restoration takes place through restoration of the geophysical environment—hydrology and associated geomorphology—in the proper ecosystem context and connectivity to assure that both the biologically desirable species and their supporting ecosystem are restored for their direct and indirect nonuse value.



One interpretation of the ecosystem restoration study objective is that ecosystem support or habitat alone may be justified based on some indication of their scarcity other than the species they support (e.g., fractions of a habitat or ecosystem type altered). No federal laws specifically establish native species variety or biodiversity as their objective, just as no federal law establishes ecosystem support or habitat as an objective independent

of what is to be supported. On the other hand, a public desire for more or less of many categories of species is expressed in the goals of numerous fish and wildlife, pest control, and other laws. Many of these laws are motivated by the desire for recreational and commercial use of wild species and protection of economic welfare, thus disqualifying them from ecosystem restoration objective consideration in the Corps. Other laws pertain to a mix of use and nonuse value, the onus being on planners to be sure of the desired outputs and how the objectives of the law pertain to them.

However, a subset of species addressed in federal law definitely identifies nonuse value for recovery. They are species that are desirable in greater quantity based on *biological* criteria (such as the *biological* criteria used to list threatened and endangered species under the protections of the ESA) rather than on economic, ethical, or other criteria. The desirability of species viability indicated in the ESA is a reflection of public “demand” for the biological importance of distinct species, not only in potential resource development but in contributing to the biodiversity that sustains ecosystems. An objective that focuses on the biological desirability of species and their restoration to a sustainable state is consistent with Corps policy positions on environmental and ecosystem sustainability.

Depressed viability indicates an insufficient supply with respect to demand based on nonuse values. In that interpretation, environmental benefits accrue as demand is satisfied with increased viability of populations indicated by age structure, sex ratio, fecundity and other indicators of population stability. This is consistent with the species viability goal of the ESA, but does not have to be limited to listed species. The ESA heritage maintenance goal is not limited to listed species; species listing and management is the authorized means for achieving the goal. Listing is generally regarded as an act of last resort when all else fails (e.g., NCR 1995). Many more species than are now listed under ESA protection are considered vulnerable and imperiled (Scott et al. 2006) and their restoration could qualify as a nonuse benefit based on the species viability criterion.

Corps planning regulations do not explicitly state that restoring unusual biological characteristics of species components of ecosystems is what is especially important for objective achievement, as it does in the WRC (1983) definition of ecological attributes. Planning guidance does nonetheless

indicate the importance of “documentation on the relative scarcity of the resources,” which helps “to determine the significance of the resources to be restored.” (USACE 2000). The planning policy regulations also more specifically identify resource scarcity as important in several indicators of resource significance. Emphasis is placed on habitat scarcity and connectivity that is scarce with respect to the needs of species. The scarcity of characteristics is an important aspect of unusual characteristics. Distinctiveness is only indicated in the planning policy emphasis on native biodiversity; it is about variation in form and function that, in turn, is about distinctiveness. The policy guidance is not clear about how habitat scarcity, connectivity and plant and animal diversity are related to one another and to the specific desired outputs from ecosystem restoration.

The Corps has used many metrics for ecosystem restoration project planning (O’Neil et al. 2001, Stakhiv et al. 2003); most of them are based on some multiple of a habitat quality metric and geographical area, generally consistent with the Habitat Evaluation Procedure (HEP) developed by FWS during the late 1970s (FWS 1980, 1981). The large majority of habitat quality metrics is based on the assumption that a proxy for ecosystem value is indicated by the suitability and geographical area of habitat for indicator species. A problem with this approach is that the “indicator species” typically are not clearly linked to restoration of “more biologically desirable species,” — the key indicator of success in ecosystem restoration projects. There is no way that the ecosystem needs of an indicator species can indicate the ecosystem support for desired outputs without identifying those desired outputs. The chronic lack of connection of indicator to objective in feasibility studies has contributed to chronic problems in project review since program inception (e.g., Brandreth and Skaggs 2002). In addition, there is no way to tally the outputs of different projects to assess added value at the program level in a concise and easily understood metric. That led to the development of entirely different indicators of project value added to the nation, which are now used to rank projects for annual budget purposes (USACE 2009).

Project ranking for maximum value added at the program level was in response to the Government Performance and Results Act. In the FY 2008 budget process as described in USACE (2009), seven criteria were used. Three of the criteria addressed resource scarcity in the form of habitat and species of special status. One of the criteria addressed the risk that the project outcome would not be naturally sustainable. Another criterion, plan

recognition, addresses the scope of collaboration in the context of an officially recognized plan of some kind. Two criteria are based on the degree to which the physical environment in the project area is restored to a more natural condition. Each of the criteria is weighted for relative importance (based on unpublished criteria) and the criteria scores are summed into an index to project rank. The criteria collectively indicate factors that are believed to be important in achieving ecosystem restoration success. The index is not directly comparable to any of the metrics used in individual project planning. Differences among project and program concepts of benefits measurement may add to existing confusion over the ecosystem restoration objective.

The protection of those restoration desired outputs that justify investment is intended to sustain ecosystem viability for desired output production indefinitely into the future. The Corps has affirmed its dedication to sustaining valued environmental qualities in various other ways. It established an environmental sustainability goal in its Environmental Operating Principles (USACE 2002) and it identified environmental sustainability objectives in the Civil Works Strategic Plan for projects planned and constructed to contribute to national economic development and for Corps owned lands (USACE 2004).

5 Measuring Nonmonetary Value Outside Federal Agencies

The environmental organizations outside federal government fall into several categories within government and NGOs. Among the NGOs are those that primarily promote information development and dissemination, those that ensure environmental laws are enforced, and others that invest membership donations in environmental land and water conservation.

Other Government Agencies

Outside of federal government, numerous state and foreign governments have interest in investing public revenues in environmental value and often work closely with NGOs with similar missions. Many of these agencies perform as regulatory or land and water resource management agencies at state and local levels, and in other nations. For the most part, like the federal agencies, they follow institutional goals, authority, and cost-effectiveness models. And, like the federal agencies, a wide array of performance indicators are used in addition to economic indicators. No attempt is made here to describe these in more detail.

Professional Scientific Organizations

All environmental NGOs provide some information to promote their mission, but the professional scientific “societies” make information provision itself a high value mission that is conducted through research and publication. Three high-profile examples are the Wildlife Society, the American Fisheries Society and the Ecological Society of America. The professional scientific organizations typically have memberships made up primarily of scientifically trained professionals including academics, researchers outside academia, and resource managers. Most of these organizations avoid extensive entanglement in values-based advocacy. They advocate primarily for scientific research and education, and sometimes disseminate position statements on issues based primarily on the science pertaining to the issue. The value of scientific information is determined by its application to use (use value) or the options it provides for future use (nonuse value). Quantification of value added is difficult and rarely attempted.

Regulatory NGOs

Regulatory NGOs operate primarily through the public oversight opportunities provided in existing and potential environmental law. They have played key roles in lobbying for more effective law and regulatory process. Much like the legislative goals they helped establish, they are motivated to protect a diverse and undifferentiated mix of use and nonuse values associated with the natural and cultural resources of the environment. Examples are The Sierra Club, National Wildlife Federation, and Environmental Defense Fund. In recent years, the strategy of the regulatory NGOs has shifted away from the generation of new law to assuring that existing laws are sustained and enforced. Much of their influence derives from authority provided in environmental law to sue government for non-performance. In that role, they generally assume, as the agencies typically do, that the benefits of enforcing the laws justify their investments of time and money. Investment decisions are largely in response to the actions or inaction of Congress and the agencies—i.e., in defense of already institutionalized environmental protection. The value they add is indirect, through the added effectiveness in goal achievement that they encourage from Congress and the agencies.

Conservancies

The land and water conservation NGOs, or conservancies, have long been concerned with returning nonuse benefits from conservation area protection (the predominant activity) and restoration that justify their investments. They have done more than other groups in and out of government to develop criteria to guide cost effective investment. Like the goals of many laws governing agency decisions, the missions of many conservancies do not separately target use and nonuse value, but a few of the largest largely target the protection of ecological resources based on their nonuse value. Among the largest and most influential of the privately funded conservancies are The Nature Conservancy (TNC), the World Wildlife Fund (WWF), and the National Fish and Wildlife Foundation (NFWF). The missions of TNC and WWF are directed largely at nonuse value that results from biodiversity protection. The mission of the NFWF is directed at both (improving recreational use is an important part of the mission). Most conservancies are privately funded, but often work closely with government agencies having similar missions and often leverage government spending to pursue their missions, some more than others. The NFWF, for example, is private but created under charter by Congress, gains

some fiscal support from Congress, and works especially close with federal agencies. Especially in the international arena, some conservancies are almost entirely government supported. Australia and South Africa, for example, have been particularly prominent in developing criteria for ranking areas for protection in biodiversity reserves.

The conservancies of most interest in this review have biodiversity protection missions and make investment decisions largely independent of present use value based on scientific knowledge of resource status. In the United States they are frequently aligned with the Society of Conservation Biologists and secondarily with other biological societies oriented toward taxonomic and ecological specialties. In the 1990s, the growing success of the biodiversity conservancies, based largely on “market solutions” (buying private properties), may have contributed to a failed movement to develop biodiversity legislation that would more proactively conserve species before they decline to the dangerously low numbers which justify ESA protection (e.g., Orians 1993). The remainder of this section describes the many approaches taken by conservancies to solving the problem of sustaining biodiversity through protection of existing resources and, less commonly, through ecosystem restoration.

Conservancy Values

The values that motivate development and achievement of biodiversity conservation objectives are complex, but their importance is widely recognized at global to local levels. The United Nations Convention on Biological Diversity in 1992 promoted national biodiversity planning in conjunction with other environmental and development plans with the intent of recovering threatened and endangered species, restoring degraded ecosystems, and protecting areas of outstanding biodiversity value, among other actions. Certainly much of the motivation to redirect some of the monetary value of biological resources, such as pharmaceuticals, from wealthy nations to underdeveloped nations was behind this international movement (Noss and Cooperrider 1994). But heritage was involved as well, based on recognized obligations to provide long-term opportunities to future generations of people worldwide.

The international biodiversity movement was closely aligned with the international sustainable development movement (WCED 1987), which was directed at sustaining present and potential use of resources for present and future generations. Protecting species from extinction is an important

aspect of sustainable development (WCED 1987). One commonly held international justification for biodiversity conservation reflected the perceived importance of maximizing present use and benefits from biological resources (e.g., commodity production, recreational tourism) while sustaining all potential use for future generations of people (e.g., Reid and Miller 1989). The parallel development of this concept with concepts of environmental sustainability became clear during the 1990s.

Goodland (1995) and Goodland and Daly (1996) developed a concept of natural capital central to the definition of environmental sustainability as it pertained to achieving sustainable development—a global goal embraced by the United Nations and the World Bank. Natural capital is made up of the natural resources that must be maintained to sustain opportunities for future natural resource use. Folke et al. (1996) and others considered biodiversity metrics, such as species number and distinctiveness, to be appropriate measures of the natural capital that is needed to sustain all known and undiscovered natural services. A broader view of natural capital also includes natural geological resources and the services they provide.

Noss and Cooperrider (1994) described categories of value that motivate biodiversity conservation and maintenance of natural capital in the sense of Folke et al. (1996). The categories include: 1) direct utilitarian value; 2) indirect utilitarian value; 3) recreation and esthetic value; and 4) intrinsic, spiritual, and ethical values. In the classification scheme of NRC (2005), at least the first three of the four categories are instrumental value, which can be bartered. The NRC (2005) included recreational and esthetic value with direct use value. The separate categorization of recreational and esthetic use is consistent with the assumption that these are in large part non-consumptive uses and that direct use implies consumptive use. The difference is important to conservation biologists because non-consumptive use is compatible with setting aside conservation areas for their nonuse value. Curiously, Noss and Cooperrider (1994) did not explicitly identify nonuse value, including heritage value, in any category. The fourth category includes intrinsic, spiritual and ethical values, which cannot be measured in units of exchange because they cannot be traded in a market or a proxy market setting.

The summary of conservancy motivations by Perlman and Adelson (1997) included both use and nonuse values. The conservancies that most clearly

invested in nonuse values of natural resources make it their mission to protect all native biodiversity, not just the minority that has direct-use value. Recent concern has been expressed that conservancies are not investing enough to sustain economically valued ecosystem services (e.g. Daily 1997), such as the pollution control, flood protection, and commercial fish production service of salt marshes (Kareiva and Marvier 2003, Molnar et al. 2004). Biases in past conservation priorities may indicate that use value has had a significant influence all along. This shows up in the lower value assigned to freshwater biodiversity and small species than to terrestrial biodiversity and large species (Abell 2002, Cole 2009). Terrestrial biodiversity and large species are more likely to be used for commodities, recreation, and aesthetic satisfaction.

Noss and Cooperrider (1994) distrusted an economic basis for justifying investment in biodiversity protection because the relationships between ecosystems and services were often difficult to determine and all value could not be captured in monetary terms. Apparently, rejecting the impracticality of it as indicated by positions argued in Norton (1986), Noss and Cooperrider favored using an ecocentric, ethics-driven concept of benefit based on the intrinsic value held in nature independent of economic considerations. Others place emphasis on the natural heritage value of biodiversity, which is clear in the title of a landmark publication, *Precious Heritage, The Status of Biodiversity in the United States* (Stein et al. 2000). Their position has remained influential in conservancies even with the recent trend to seek monetary valuation of natural goods and services to show certain sectors of their memberships the use value of conservancy actions (Pagiola et al. 2004).

Regardless of the underlying membership motivations for investing in conservation-area protection, it offsets destructive use of resources and sustains natural resource heritage. The strategy is consistent with the concept of sustaining natural capital for long-term environmental sustainability (Goodland and Daly 1996), for which the prevalent investment justifications are sustained resource use and maintenance of natural heritage.

Conservancy Planning

Groves (2003) has summarized much of what has transpired in biodiversity conservancy planning over recent decades. Well-documented processes were published for TNC in 1997 (TNC 1997, Groves et al. 2000) and for the

WWF in 2000 (Dinerstein et al. 2000). Taking planning processes to the public was motivated in large part by the perceived need of these and other large biodiversity conservancies to integrate and collaborate more with each other and the many more smaller conservancies because of limited resources and depressing trends in biodiversity status worldwide.

The basic strategy of most conservancies has been to acquire private properties and property easements in an intact natural state to achieve the objectives of the conservancy. They also seek to restore desired natural value when necessary, and limit any use that devalues intended utility by the conservancy. These strategies are similar to public land acquisition strategies for conservation purposes. The criteria used to select conservation areas are indicators of value perceived in conservation benefit (this will be described later). To the extent they are selected to protect nonuse values, they are also indicators of nonmonetary benefit as understood by the Corps of Engineers.

The conservancy planning process has been profoundly influenced by the rapid improvement in computing, remote sensing, and geographical information systems (GIS) software technology, and by the need for making conservation area investment decisions in many locations with little existing biodiversity data. The ecoregion concept is increasingly used to organize data on species, communities, ecosystems and environmental factors into more manageable units of ecological variation within continental contexts. This strategy relies largely on underlying scientific concepts of island biogeography (MacArthur and Wilson 1967), species-area relationships (e.g., Rosensweig 1995), and landscape ecology (e.g., Turner et al. 2001).

Among possible regional approaches, ecoregions are preferred because ecoregional boundaries are discernable from existing maps and satellite imagery, and are much better than political boundaries for delimiting species distributions. Numerous ecoregional classification schemes have been developed, none of which works uniformly well for all conservancy purposes (Groves 2003). For the United States, TNC has used Bailey (1995) for terrestrial ecosystems and a new national vegetation classification developed by the US Department Agriculture, Forest Service (Groves 2003).

Johnson (1995) developed conservation planning principles that emphasized the need for clearly stated conservation goals linked to local,

national and global priorities, and to a clearly expressed priority-setting process pointed toward goal achievement. He advocated that alternative priority setting procedures be examined, compared for their effectiveness, including how priorities fit into policy and institutional contexts, and reassessed and revised at regular intervals. He pointed out the importance of using all relevant information, including input from all stakeholders who can have an impact on effectiveness. The first principle emphasized the importance of goal and objective setting. The remaining principles were primarily strategic and tactical approaches to effective achievement.

Groves (2003) built on these principles to follow steps common to most planning protocols, but couched in activities specific to conservancy decision needs. The steps in general include specifying objectives and following strategies to achieve the objectives within each ecoregion. The strategies include inventorying existing information, using that information to formulate different conservation area system plans for objective achievement, evaluating the effectiveness of each plan, comparing plans, and selecting the best plan. This approach is much like government approaches to planning. Goals and objectives are identified and measures are taken to achieve them. The primary difference is that the goals and objectives of private conservancies are not imposed by the public through Congress or the President, and the measures taken are not limited to authorities granted by Congress.

Conservancy Objectives

Conservation biologists in the conservancies and academia have done much research and analysis in pursuit of protocols for ranking the choice of conservation areas to most cost-effectively implement a conservation area network in each ecoregion as needed. Numerous criteria have been suggested and used to select conservation areas related to the conservation objectives and strategies used to achieve them.

The long-term objective of biodiversity conservancies is to secure biodiversity from net loss—to assure sustainability of naturally evolving, genetically distinct populations, species, communities and ecosystems. In conservation area network planning, objectives are usually identified as conservation targets (Groves 2003). In the short run, the high priority targets are distinctive and irreplaceable biological features of biodiversity most at risk of permanent loss because of human impacts. The targets range from sub-specific to ecosystem-level indicators of biodiversity, but

the most fundamental unit of concern for biodiversity planning is to prevent the loss of genetically unique populations with unique attributes. Historically, conservancies have focused on populations through protection of species and supporting ecosystems. They purchase properties and easements at various scales that are consistent with the need to sustain one or more populations (Noss and Cooperrider 1994).

The criteria developed to guide conservancy investments reveal much about the objectives and strategies of conservancies. From a review of the early conservation literature, Margules and Usher (1981) determined that one or more of five criteria were most often used to rank the value of wildlife conservation areas. They include the diversity, rarity, naturalness, and geographical area of the conservation area gained by the investment, and the threats that impose risks on the sustainability of the investment returns. Margules and Usher (1981) found that the order of importance placed on the five criteria varied widely among conservancies. Nevertheless, subsequent plans and analyses substantiate the consistent importance of all five criteria in biodiversity conservation planning.

The focus of the of TNC planning process expresses an objective in more elaborate terms: “ensure that the world’s species, communities, and ecosystems, and the underlying ecological processes that sustain them, will not only persist, but continue to evolve and adapt for generations to come” (Groves 2003). The central objective is to sustain an ecological and evolutionary process that continuously regenerates diversity in the ecoregion. This objective is consistent with earlier assessments of the scientific basis for biodiversity conservation (e.g., Reid and Miller 1989). Because evolution occurs at the population level, the objective of TNC is population-centered, but achieved through protection of populations in the context of supporting ecosystems. Biodiversity in all of its forms is assumed to be sustained if representative examples of each population can be sustained in a naturally adaptive state where populations interact with one another and with other aspects of their environment. While the prevalent strategy has been to approach this objective through conservation of ecosystems consistent with the information available, some conservation biologists are recognizing limitations of ecosystem approaches using gross indicators of boundaries in the absence of good species-population data (Brooks et al. 2004).

The population focus of the conservation concept is consistent with the emphases of conservation and evolutionary science. That science has and will continue to advance most through incremental study of individual populations and through community and ecosystem studies that are limited to local populations. The concept of species is most often developed through study of individuals and populations, only sometimes including measures of the variation among them. Conservation science has been organized primarily at the level of species distributions and other measures of species status in an attempt to generalize understanding developed at the individual and population level. While the TNC focus indicates the ultimate goal of biodiversity conservation (the continuity of natural evolutionary process) and the general strategy for achievement, it does not identify specific objectives.

Dinerstein et al. (2000) emphasized the conservation of biological distinctiveness as an objective of conservation area selection within ecoregions. Distinctiveness is indicated by the endemic taxa, rare communities, species richness, and “unique ecological or evolutionary phenomena.” These indicators of distinctiveness are species-population oriented, including the rare communities, which are rare largely because the populations comprising them are rare. By this definition, rarity is included in the concept of distinctiveness, but two species of similar rarity can differ greatly in the distinctiveness of the attributes. One may be the only member of its taxonomic family with many unique attributes while the other may have many closely related species with all but a few attributes in common. Loss of the highly distinctive species is likely to take with it many more unique attributes than the loss of any one of the closely related species.

Threats to populations and species can be incorporated into a rarity-based objective by translating them into predictions of future rarity. For two populations of past similar abundance, the declining population is rarer than the stable population, considering the time it takes to implement plans. The priority rests with the population of lower predicted rarity. Other criteria identified by Groves et al. (2000) and Dinerstein et al. (2000) pertaining to the objectives of conservation include the degree that conservation targets have already been conserved elsewhere and the number of rare or endangered species. Both of these are refinements of an objective based on securing the viability of rare populations.

Noss and Cooperrider (1994) emphasized prioritization of conservation areas with high species richness and high frequency of species found nowhere else (endemism) in ecoregions with little other available information; this condition is common in many parts of the world. Species richness counts the number of distinctive forms in the prospective conservation area and endemism identifies the number of those species that are uniquely limited to the area. Both are indicators of the number of distinct and rare components in ecosystems. Areas of high species diversity usually have higher numbers of both common and rare species, which are often endemic species limited to small ranges. Geographically small ecosystems with high diversity and endemism have especially distinctive and rare species.

With few exceptions, the species considered as conservation targets summarized in Groves (2003) have one thing in common: the number of secure populations at some point in the future is predicted to be too low to sustain population and species viability. With enough information about species viability and anticipated changes in viability, conservation areas might be prioritized and selected based on the projected time of functional extinction. Rare species are inherently more vulnerable to extinction because they exist in numbers that provide little protection from any threats that materialize. The general rule has exceptions, however, as revealed by harvest depletions of very abundant species; some of them to extinction.

The most relevant indicator of biodiversity conservation objective achievement is long-term viability of all species in self-regulating ecosystem and evolutionary settings. Biodiversity protection benefits accrue with increased protection of species populations from extinction. Consistent with this thinking, the conservation status of species designated in the NatureServe Explorer database (NatureServe 2009) is based on the number, distribution and viability of species populations. NatureServe explorer was developed by TNC and the state natural heritage programs and is now operated by NatureServe, an independent NGO. The database is widely recognized and used by government and nongovernment conservation agencies and organizations.

Program objective achievement is indicated by assurance of naturally sustained species viability in all ecoregions. This is the same program objective espoused in the ESA. The strategies, however, are less focused on

protection and recovery of threatened and endangered species listed under ESA protection (Groves 2003) than on concern for all species showing signs of vulnerability, regardless of listed status. In the United States, these number several times the number of species listed under ESA protections.

Conservancy Strategies

Strategic criteria largely address the management of risks that threaten the success of plan implementation and program success. Strategic criteria greatly influence specific conservation area and plan selection with the objective of securing all biodiversity, with priority placed on rare and distinctive biodiversity. The difficulty of doing this piecemeal to achieve program goals is increasingly recognized; collaboration, including the integration of planning process, is increasingly championed as a master strategy (Groves 2003). This has been among the motivations for recent publications of planning process among TNC, the WWF and a few other conservancies.

Obtaining full and sustainable representation of an ecoregion's biodiversity in conservation area networks is essential if the risks to biodiversity viability are to be effectively managed. Making use of all reliable information to accomplish this is one of the fundamentals (Johnson 1995). Representativeness includes all genotypes, species, ecosystems, and landscapes in a network of reserves (Noss and Cooperider 1994, Groves et al. 2000), but fully representative protection requires attention to all ecoregions. This is a fundamental reason why some conservancies have gone global in perspective.

In managing the risk of not including all biodiversity in a protected status, the WWF places great store in landscape integrity, which is indicated by large blocks of intact (non-fragmented) ecosystem with high internal connectivity (Dinerstein et al. 2000). In the nascent stage of conservation planning, Diamond (1975) developed broad strategies for managing the risks of not selecting representative and sustainable conservation areas. He expressed these strategies as principles for animal conservation, but they amounted to criteria for selecting conservation areas. The ideas were based on island biogeography concepts developed by MacArthur and Wilson (1967) and others who subsequently followed. These criteria included selecting 1) large areas over smaller ones; 2) a single area over separate smaller areas of similar total size; 3) areas closer together over those farther

apart; 4) clustered areas over those lined up in a row, 5) corridor-connected areas over isolated areas; and 6) round areas over long, thin ones. These influenced the similar criteria of Noss and Cooperrider (1994) and Noss et al. (1997).

For the most part, other strategies address the risk that the biodiversity in the selected conservation areas will not represent all biodiversity in the ecoregion. Pressey et al. (1993) emphasized flexibility when considering many potential conservation area arrangements when formulating network plans that can be compared for their effectiveness. They also emphasized that each conservation area in the planned network should be evaluated for its “complementarity” in contributing to achievement of ecoregional biodiversity objectives; in other words, each conservation area must add value by significantly adding to the security of ecoregional biodiversity (Davey 1998). Pressey et al. (1993) also emphasized careful assessment of how essential a candidate conservation area is to securing ecoregional biodiversity. A candidate conservation area with many equally protective potential substitutes is much less critical to include in the conservation network than one for which there are no substitutes. A conservation area is least substitutable when it includes the only known remnants of rare and distinctive biodiversity.

The cost-effectiveness of conservation-area plan selection also is an important strategic consideration (Davey 1998, Ando et al. 1998, Polasky et al. 2001, Newburg et al. 2005, Wilson et al. 2006). Spending more than is necessary on conservation network implementation in any ecoregion increases the risk that inadequate funding will limit objective achievement. Generally speaking, conservation areas needing extensive restoration are ranked low when intact natural areas are available, due to the higher costs, delayed protection, and risks associated with restoration failure (Dinerstein et al. 2000). Dinerstein et al. (2000), Groves et al. (2000) and Groves (2003) limited restoration recommendations to improvement of an essential but degraded conservation area and to creation of conservation area redundancy in ecoregions that have been largely converted to human use. Some terrestrial ecosystems and numerous aquatic ecosystems fall into these categories.

Reducing the risks that threaten the ecosystem integrity of conservation areas is also a key to success. Noss and Cooperrider (1994), Davey (1998), Margules and Pressey (2000), and Groves et al. (2000) emphasized the

need for the natural processes within a conservation area to sustain the rare and distinctive elements of biodiversity far into the future without reliance on human discretion and management action. This strategy requires conservation areas and networks of appropriate sizes and arrangements to facilitate natural adaptation to natural events—such as hurricanes, floods, storms, and droughts. Much has been written about criteria for the design of conservation area shape, size, and connectivity (see Noss and Cooperrider and Groves 2003 for reviews) since MacArthur and Wilson (1967) and Diamond (1975) established basic principles pertaining to relationships between biological diversity and geography.

Shaffer and Stein (2000) and Groves (2003) added resilience and redundancy to criteria used to plan conservation area networks. In their view, each conservation area and the entire network of areas should exhibit an ecosystem integrity that facilitates rapid recovery from local human and natural disturbances. The networks should provide the redundancy necessary to sustain biodiversity in areas where individual areas are prone to disturbance by humans or natural events.

Gap analysis is a strategy developed to manage insufficient information for cost-effective selection of conservation areas (Jennings 2000). Gap analysis is an approach to evaluating gaps in biodiversity representativeness among nature reserves. It relies on overlays of data pertaining to the distribution of species, their habitats, and threats to habitat continuity, and on analyses using Geographical Information System software. The Department of the Interior, US Geological Survey manages the Gap Analysis Program (GAP), which is a nationwide attempt in the United States to manage biodiversity information deficiencies (Jennings 2000). Gap analysis is also currently used to identify hotspots by the WWF, Conservation International, and others to establish global priority settings.

Managing the threat to conservation areas and area networks is critical, especially when conservation areas are essential, and threats are ecoregionally pervasive and immediate. Margules and Pressey (2000) developed a systematic approach to conservation planning that focuses on threats to potential conservation areas that increase the risk of them being “transformed by extractive uses.” Dinerstein et al. (2000) measured the severity of threat by the rate and extent of anticipated effect throughout the conservation area under consideration.

Hunter (1991) and Groves (2003) advocated using both coarse- and fine-grain approaches to conservation area planning to better manage threats. They recommend considering both ecoregional distinction in biodiversity at the ecosystem scale and species distinction and scarcity at the population and species distribution scales. Information needs are similar to those for gap analysis: ecosystem, community, and species distributions and conditions; land ownership and use patterns; and trends relevant to both. When there is more specific information about the occurrences of species populations and other conservation targets, those areas with the greater diversity and number of occurrences are usually ranked higher (Groves et al. 2000).

Dinerstein et al. (2000) and Groves et al. (2000) also considered strategies for leveraging more investment interest by highlighting high profile conservation targets with the intent of conserving habitat for many species associated with them. The targets are often charismatic “flagship” species useful for gaining public support (e.g., whooping cranes and wolves) and other “umbrella” species. Umbrella species are frequently large species with large ranges over environments mostly free of much human impact. “Focal” species make up a set of species with collective ranges that have the same effect as an umbrella species when conserved (e.g., Noss et al. (1999). Habitat conservation for “keystone species” also has outsized effect on species conservation because other species functionally depend on them, (e.g., Soule and Terbough 1999). Common “indicator” species may sometimes be used to indicate the needs of threatened species (Groves 2003), but the conservation value accrues with the conservation of the most threatened species, not the common indicator species.

Like many past approaches to natural resources planning, conservation planning based on setting aside spatially fixed conservation areas assumed stationarity in the suitability of the habitats set aside within them. Adaptive management has been dismissed as inappropriate because it was thought to permit too much opportunity to make bad decisions for preserving biodiversity; second, there appeared to be no going back once species are lost (Groves 2003). More recently, however, recognition of the potential effects of major changes in the landscape has led to some strategic thinking about what to do (Pressey et al. 2007). One means is to use variable representation targets for conservation areas, which can be increased or decreased as needed. Another makes use of movable conservation areas that can be adjusted spatially as conditions demand. The design criteria for

conservation area networks can be modified to allow for species' range change adaptations. Another strategy is to set aside areas that show potential for sustaining species through periods of stress and flux. Issue management is technically and economically daunting in regions that have been largely converted to human use on privately controlled properties of high value to the owners. These same issues face ecosystem restoration programs with similar goals.

Some attempts have been made by the conservancies to include strategic criteria for conservation area selection in formal ranking systems and selection algorithms. A wide variety of scoring methods have been developed to prioritize conservation area selection based on the criteria that have already been described (Kirkpatrick 1983, Smith and Theberge 1986, Usher 1986, Margules et al 1988, Pressey and Nichols 1989, Dinerstein et al. 2000 and Groves et al. 2000). Most of these scoring methods are based on species-level information and tend to be qualitative. Certain scoring methods too often favor the repeated occurrences of a few conservation targets that most meet the criteria used, causing other deserving species to be overlooked. The use of scoring methods has decreased with the recent trend toward emphasizing ecosystem selection over species-based selection in many nations where species information is scarce. Other approaches rely on statistical techniques to predict the distributions of conservation targets from historic data (e.g., museum records, maps) , newly acquired data (e.g., Ferson and Bergman 2000, Vaughn and Ormerod 2003) or from an understanding of species' needs (Araujo and New 2007). All of these approaches have limitations associated mostly with the amount of information available.

Conservancy Effectiveness

Much like government, reports of conservancy effectiveness are based more on process than on results, and this, of course, bypasses direct measurement of benefit. A common way to report progress (and indicate benefit) is in conservation area protected by purchase and easement or other agreement and periodic verification that protected ecosystems actually remain intact. The implicit assumption is that biodiversity protection is proportionally more than the area of ecosystem protected based on conservation area selection criteria. The actual extent to which progress is being made in sustaining biodiversity is less clear. The issue depends largely on the rate at which data on projected species scarcity are updated. Whether that is done at the species or the ecosystem level,

appropriate indicators of success are needed. That is complicated by limited scientific understanding at both levels, but especially at the ecosystem level (Groves 2003).

Some have argued that not enough species data (fine-grain data) exist even in places like North America to justify area selection at that scale (Franklin 1993) and that the costs of getting such data when it is not already available may exceed the benefit. Consistent with that view, Groves (2003) argues that the coarse-grain approach includes species that would not be otherwise included. These positions -- while strategically wise -- do little to address the need for monitoring performance effectiveness, which has been called into question due to procedural issues and the question of whether all biodiversity is captured in the approach.

Species-based approaches have certain advantages when the information is available. Unlike bio-taxonomy, a field in which one classification scheme is well-understood with respect to its limitations (and is universally established and accepted among biologists), many different classifications exist for communities and ecosystems (e.g., vegetation type, plant associations, animal assemblages, dominant species, geography, climate (Groves 2003)), and their limitations are not always wellunderstood. For that reason, more attention must be paid to a clear definition of intended targets when communities and ecosystems are used (Noss et al.1995) and what the protection of the ecosystems means in terms of value added. Ecosystems vary widely in the number of unique attributes, which in any particular space change through time. These uncertainties have an uncertain effect on how well the prioritization of conservation areas maximizes diversity preservation. The issue is magnified in ecosystems that change along a continuum, as many river and coastal systems do.

Conserving a geographical area based on ecosystem boundaries defined by climatic, physiographic, or plant-structure indicators often misses significant biodiversity without detailed species information available to assure otherwise (Brooks et al. 2004, Arponen et al. 2005). Ferrier and Watson (1997) inferred from their study that the species approach is most dependable where data are available. They found that the completeness of native biodiversity within mapped ecosystem boundaries was better indicated by models of species distributions than by physical attributes of the environment or by vegetation classification. Pressey (2004) expressed a more comprehensive and idealistic view by recognizing a need to

assemble the best data at different scales to achieve conservation objectives using complementary approaches. But he also emphasized the importance of species data in the mix.

Data availability for species has increased markedly in developed nations because of the efforts of conservation organizations. NatureServe Explorer, for example, provides in-depth coverage for many species that can serve as indicators for community and ecosystem condition in the United States. These data continue to accumulate rapidly for some taxonomic groups, but the process is based on limited funding that competes with investments in conservation areas. A review of the database quickly reveals that the intensity of monitoring is unevenly distributed and favors species valued for esthetic, recreational, commercial, or other economically measurable value.

Data on species distribution and conservation status -- when available for a large number of indicator species -- provide more information for characterizing changes in the scarcity and distinctiveness of communities and ecosystems in particular parts of an ecoregion than does a map with environmental, vegetative or other more general indicators of distinctiveness and rarity. Care must be taken with the use of indicator species, however, since they may not be good indicators of all biodiversity conditions (Groves 2003). Nevertheless, there seems to be no better way of monitoring the effectiveness of biodiversity protection.

6 Conclusions

1. Environmental investment decision-making by government agencies is based on resource utility value added by protection and restoration. It requires tradeoffs to determine the best investment of tax revenues and donations. The utility value is derived from resource use and from setting aside use in a “nonuse” status.
2. Most, if not all, use value can be measured in monetary terms without controversy whereas nonuse value cannot be measured in monetary terms without controversy. The Corps of Engineers has prohibited the use of controversial stated preference techniques for measuring nonuse value in project planning.
3. Environmental value is typically established in the goals and objectives of government legislation and NGO missions, which often include an undifferentiated mix of use and nonuse values. Goal and objective achievement is typically measured using nonmonetary performance indicators that do not sort use from nonuse value and offer little for measuring achievement of objectives focused on either use value or nonuse value alone.
4. In its concept of environmental value maintenance, NEPA implicitly differentiates between use value of resources that can be destroyed but replaced with substitutes, and nonuse value associated with irreplaceable resources that must be protected from destruction. The nonuse value is characterized as natural and cultural heritage.
5. Unlike most laws, the goal of the Endangered Species Act (ESA) is largely motivated by the nonuse value associated with sustaining the viability of all but a few plant and animal species in a national heritage. Congress directed the agencies administering the ESA to determine the listing of species under ESA protection based on biological criteria alone, regardless of use value.
6. The EQ improvement authority of the Corps of Engineers is unique in two ways: first, it applies to EQ improvement through ecosystem restoration and protection in virtually any aquatic public setting, and not from the authority of public land ownership; second, it must (according to policy) justify its public investments based on value that is other than the monetary value of resource use. By deduction, the value justifying investment is nonuse value.

7. The nonuse value most relevant to EQ improvement accomplished by the Corps is natural heritage value. This conclusion derives from the national heritage maintenance outcome determined for the water resources EQ objective in early federal water resources planning policy; EQ protection as determined in current policy guidance and its heritage emphasis; the heritage goals of NEPA, ESA and other law; and from the ecological resource focus of the Corps' EQ improvement authority.
8. The goal of biodiversity conservancies is to protect (and restore only when necessary) ecosystems with the intent of naturally sustaining all species, including variation within species, independent of their use value. Priorities have been set using many different variables, but relative ecosystem scarcity (security), community distinctiveness, cost, and residual risk of failure are among the most consistent. While the conservancies may sometimes favor "flagship" species with high use value, the action is intended to leverage support for a nonuse goal.
9. The goals of the biodiversity conservancies, the ESA, and the Corps ecosystem restoration program have much in common in that they largely direct the restoration or recovery of "biologically desirable species" through ecosystem support, when appropriate, and for nonuse values independent of use value measured in monetary terms. The nonuse value is most associated with the natural heritage left to future generations.
10. Species and ecosystem viability goals are challenging to meet because the projected costs are substantially greater than existing funds; thus, cost-effectiveness through improved government and NGO collaboration is essential. Collaboration may be improved by the use of measures of effectiveness that can be easily compared across programs.
11. Commonly used criteria for nonuse value culled from NEPA, the ESA, and from the biodiversity conservancy literature include the security of ecosystem elements (e.g., species) from extinction, the distinctiveness of the elements, the costs of not achieving individual element sustainability within intact ecosystems, and the unmanaged risk that the investment will fail to produce intended results.
12. The commonly used criteria are similar to the scarcity, diversity, viability, and cost-effectiveness criteria identified in Corps planning policy as important considerations of ecosystem restoration investment justification and they appear to be the basis of a comprehensively useful nonmonetary metric for restoring and protecting nonuse natural heritage value.

References

- Abell, R. 2002. Conservation biology for the biodiversity crisis: A freshwater follow-up. *Conservation Biology* 16: 1435-1437.
- Ando, A., J. Camm, S. Polasky, and A. Solow. 1998. Species distributions, land values, and efficient conservation. *Science* 279:2126-2128.
- Apogee Research, Inc. 1996. Monetary measurement of environmental goods and services: Framework and summary of techniques for Corps planners. IWR Report 96-R-24. Alexandria VA: US Army Institute for Water Resources.
- Arponen, A., R. K. Heikkinen, C. D. Thomas, and A. Moilanen. 2005. The value of biodiversity in reserve selection, species weighting, and benefit functions. *Conservation Biology* 19: 2009-2014.
- Araújo, M. B., and M. New. 2007. Ensemble forecasting of species distributions. *Trends in Ecology and Evolution*, 22: 42-7.
- Bailey, R. G. 1995. *Description of ecoregions of the United States*. US Forest Service Miscellaneous Publication Number 1391. Washington, DC: USDA Forest Service.
- Barbier, E. B. 1994. Valuing environmental functions: Tropical wetlands. *Land Economics* 70:155-173.
- Barbier, E. B. 2000. Valuing the environment as input: Applications to mangrove-fisheries linkages. *Ecological Economics* 35:47-61
- Barbier, E. B., G. Browns, S. Dalmazzone, C. Folke, M. Gadgil, N. Hanley, C. S. Holling, W. H. Lesser, K.-G. Maler, P. Mason, T. Panayotou, C. Perrings, R. K. Turner, and M. Wells. 1995. The economic value of biodiversity. Pages 822-914 In V. H. Heywood ed. *Global Biodiversity Assessment*, New York, NY: Cambridge University Press.
- Barbour, M. T., J. Gerritsen, J. S. White, and M. L. Bastian. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 15(2):185-211.
- Bartoldus, C. C. 1997. A comprehensive review of wetland assessment procedures: A guide for wetland practitioners. St. Michaels, MD: Environmental Concern Inc.
- Bateman, I. J., R. C. Carson, B. Day, M. Hanemann, N. Hanley, T. Hett, M. J. Lee, G. Loomes, S. Mourato, E. Ozdemiroglu, D. Pearce, R. Sugden, and J. Swanson. 2002. (Economic valuation with stated preference techniques: A manual.) Cheltenham, U. K. Edward Elgar.
- Bender, F. L. 2003. The culture of extinction: Toward a philosophy of deep ecology. Amherst, NY: Humanity Books.

- Bovee, K. D. 1982. *A guide to stream habitat analysis using the Instream Flow Incremental Methodology*. Instream Flow Information Paper 12. FWS/OBS-82/26. Washington, DC: US Fish and Wildlife Service.
- Boyle, K. J. 2003. Contingent valuation in practice. In *A Primer on Nonmarket Valuation*, ed. I. K. Champ, J. Boyle, and T. C. Brown, Chapter 5. Boston, MA: P. Kluwer.
- Brandreth, B. and L. L. Skaggs 2002. *Lessons learned from cost effectiveness and incremental cost analysis*. IWR Report 02-R-5. Alexandria, VA: US Army Institute for Water Resources.
- Brooks, T. M. G. A. B. da Fonseca, and A. S. L. Rodrigues. 2004. Protected areas and species. *Conservation Biology* 18: 616-618.
- Caldwell, L. K. 1998. *The National Environmental Policy Act: An agenda for the future*. Bloomington, IN: Indiana University Press.
- Callicott, J. B. 2006. Explicit and Implicit Values. In *The Endangered Species Act at Thirty: Conserving Biodiversity in Human-dominated Landscapes, Volume 2*, ed. J. M. Scott, D. D. Goble, and F. W. Davis, Chapter 4. Washington, DC: Island Press.
- Cole, R. A. 2009. *The sustainability of freshwater biodiversity and water resources development policy of the Army Corps of Engineers*. IWR Report 09-12-9. Alexandria, VA: US Army Institute for Water Resources.
- Cole, R. A. 2010. *A new nonmonetary metric for indicating environmental benefits from ecosystem restoration projects of the US Army Corps of Engineers*. ERDC/TL TR-10-12. Vicksburg, MS: US Army Engineer Research and Development Center,
- Cole, R. A. 2011. *Concept acceptability of non-monetary environmental benefits metrics for ecosystem restoration projects planned by the US Army Corps of Engineers*. Draft Technical Report. Vicksburg, MS: US Army Engineer Research and Development Center.
- Cole, R. A., T. D. Feather and J. D. Muncy. 2005. Watershed planning and management in the United States. In *Economics and Ecological Risk Assessment*, ed. R. J. F. Bruins and M. T. Heberling, 11-42. Boca Raton, FL: CRC Press.
- Cole, R. A., and F. A. Ward. 1996. Sustaining reservoir fisheries and angler benefits through comprehensive management analysis. In *Multidimensional Approaches to Reservoir Fisheries Management*, ed. L. E. Miranda and D. R. Devries. Bethesda, MD: American Fisheries Society.
- Daily, G. (Editor). 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington DC: Island Press.
- Davey, A. 1998. National system planning for protected areas. *Best Practice Protected Areas Guidelines Series No. 1*. World Conservation Union (IUCN): World commission on Protected Areas, Gland, Switzerland.
- Diamond, J. 1975. The island dilemma: lessons of modern geographic studies for the design of natural preserves. *Biological Conservation* 7:129-146.

- Dinerstein, E., G. Powell, D. Olson, E. Wikramamayake, R. Abell, C. Loucks, E. Underwood, T. Allnut, W. Wettengull, T. Ricketts, H. Strand, S. O'Connor, N. Burgess, and M. Mobley. 2000. *A workbook for conducting biological assessments and developing biodiversity visions for ecoregion-based conservation*. Washington, DC: WWF and the World Bank.
- Dunlap, R. E., and A. G. Mertig (Editors). 1992. *American Environmentalism: The US Environmental Movement, 1970-1990*. Washington, DC: Taylor and Francis.
- Ekins, P. 2000. *Economic Growth and Environmental Sustainability: The Prospects for Green Growth*. New York, NY: Routledge.
- Farber, S., R. Costanza, D. L. Childers, J. Erickson, K. Gross, M. Grove, C. S. Hopkinson, J. Kahn, S. Pincetl, A. Troy, P. Warren, and M. Wilson. 2006. Linking ecology and economics for ecosystem management. *BioScience* 56: 121-123.
- Feldman, D. L. 1991. *Water Resources Management: In Search of an Environmental Ethic*. Baltimore, MD: Johns Hopkins University Press.
- Ferson, S., and M. A. Bergman (editors). 2000. *Quantitative Methods in Conservation Biology*. New York, NY: Springer.
- Ferrier, S., and G. Watson. 1997. An evaluation of the effectiveness of environmental surrogates and modeling techniques in predicting the distribution of biological diversity. Special Report. Canberra, AU: NSW National Parks and Wildlife Service, Department of the Environment, Sport and Territories.
- Folke, C., C. S. Holling, and C. Perrings. 1996. Biological diversity, ecosystems, and the human scale. *Ecological Applications* 6: 1018-1024.
- Franklin, J. F. 1993. Preserving biodiversity: species, ecosystem or landscapes? *Ecological Applications* 3: 202-205.
- Freeman III, A. M. 2003. *The Measurements of Environmental and Resource Values: Theory and Methods*. Second Edition. Washington, DC: RFF Press.
- Furtado, J. I.R., and T. Belt, R. Jammi. Editors. 2000. *Economic development and environmental sustainability: Policies and principles for a durable equilibrium*. WBI Learning Resources Series. World Bank, Washington, DC.
- US Fish and Wildlife Service, (FWS) 1980. *Habitat evaluation procedures (HEP)*. ESM 102. Washington, DC: Division of Ecological Sciences, US Fish and Wildlife Service.
- US Fish and Wildlife Service (FWS). 1981. *Standards for the development of habitat suitability index models*. ESM103. Washington, DC: Division of Ecological Sciences, US Fish and Wildlife Service.
- Goodland, R. 1995 The concept of environmental sustainability. *Annual Review of Ecology and Systematics* Vol 26:1-24
- Goodland, R., and H. Daly. 1996. Environmental sustainability: Universal and non-negotiable. *Ecological Applications* Vol 6: 1002-1017.

- Groves, C. R. 2003. Drafting a conservation blueprint: A practitioner's guide to planning for biodiversity. Washington, DC: Island Press.
- Groves, C., L. Valutis, D. Vosick, B. Neely, K. Wheaton, J. Touval, and B. Runnels. 2000. *Designing a Geography of Hope: A Practitioner's Handbook for Ecoregional Conservation Planning*. Arlington, VA: The Nature Conservancy.
- Hanemann, W. M. 1994. Valuing the environment through contingent valuation. *The Journal of Economic Perspectives* 8(4): 19-44.
- Hays, S. P. 1957. Conservation and the Gospel of Efficiency, the Progressive Conservation Movement, 1890-1920. Cambridge, MA: Cambridge University Press.
- Hays, S. P. 1987. Beauty, Health, and Permanence: Environmental Politics in the United States, 1955-1985. New York, NY: Cambridge University Press.
- Hays, S. P. 1998. *Explorations in Environmental History*. Pittsburg, PA: University of Pittsburg Press.
- Hays, S. P. 2000. *A History of Environmental Politics Since 1945*. Pittsburg, PA, University of Pittsburg Press.
- Headquarters, US Army Corps of Engineers (USACE). 1999. Civil works ecosystem restoration policy. Engineer Regulation 1165-2-501. Washington, DC: Headquarters, US Army Corps of Engineers.
- . 2000. *Planning guidance notebook*. Engineer Regulation 1105-2-100GN. Washington, DC: Headquarters, US Army Corps of Engineers.
- . 2002. *Environmental operating principles*. Washington, DC: Headquarters, US Army Corps of Engineers, <http://www.hq.usace.army.mil/CEPA/7%20Environ%20Prin%20web%20site/Page1.html>.
- . 2004. *Department of the Army Corps of Engineers Civil Works Strategic Plan Fiscal Year 2004 – Fiscal Year 2009*. Washington DC: Headquarters, US Army Corps of Engineers. http://www.usace.army.mil/cw/hot_topics/ht_2004/cw_strat.pdf.
- . 2009. *Army Program Corps of Engineers Civil Works direct program development guidance, fiscal year 2011*. Circular No. 11-2-187. Washington DC: Headquarters, US Army Corps of Engineers. http://www.usace.army.mil/CECW/PID/Documents/fy11ec/FY11_Budget_EC_FINAL.pdf.
- Hilsenhoff, W. L. 1987. An improved biotic index of organic stream pollution. *Great Lakes Entomologist*: 20: 31-39.
- Holl, K. D., and R. B. Howarth. 2000. Paying for restoration. *Restoration Ecology* 8 (3): 260-267.
- Holmes, T., and W. Adamowicz. 2003. Attribute-based methods. In *A Primer on Nonmarket Valuation*, ed. I.K. Champt, J. Boyle, and T. C. Brown, Chapter 6. Boston, MA: P. Kluwer.

- Hunter, Jr., M. L. 1991. Coping with ignorance: the coarse filter strategy for maintaining biodiversity. In *Balancing on the Brink of Extinction: The Endangered Species Act and Lessons for the Future*, ed. K. A. Kohm, 266-281. Washington DC: Island Press.
- Jennings, M. D. 2000. Gap analysis: concepts, methods, and recent results. *Landscape Ecology* 15: 5-20.
- Johnson, N. 1995. Biodiversity in the balance: Approaches to setting geographic conservation priorities. Washington, DC: Biodiversity Support Program.
- Kareiva, P., and M. Marvier. 2003. Conserving biodiversity coldspots. *American Scientist* 91: 344-351.
- Karr, R. J., and E. W. Chu. 1999. Restoring Life in Running Waters: Better Biological Monitoring. Washington, DC: Island Press.
- King, D., and M. Mazzotta. 2002. Valuation of ecosystems. Website.
<http://www.ecosystemvaluation.org/>
- Kirkpatrick, J. B. 1983. An iterative method for establishing priorities for the selection of nature reserves; an example from Tazmania. *Biological Conservation* 2:316-328.
- Kline, B. 1997. First along the River: A Brief History of the US Environmental Movement. San Francisco, CA: Acada Books.
- Krutilla, J. V. 1967. Conservation reconsidered. *American Economic Review* Vol 57(4): 777-786.
- Lavigne, P. M. 2003. Greening the US Constitution. *Conservation Biology* 17: 1485-1486.
- Louviere, J. B., D. Hensher, and J. Swait. 2000. *Stated Choice Methods—Analysis and Application*. Cambridge, UK: University Press.
- Maass, A. 1951. *Muddy Waters: The Army Engineers and the Nation's Rivers*. Cambridge, MA: Harvard University Press.
- MacArthur, R. H., and E. O. Wilson. 1967. *The Theory of Island Biogeography*. Princeton, NJ: Princeton University Press.
- Margules, C. R., and M. B. Usher. 1981. Criteria used in assessing wildlife conservation potential: a review. *Biological Conservation* 21:79-109.
- Margules, C. R., A. O. Nicholls, and R. L Pressey. 1988. Selecting networks of reserves to maximize biological diversity. *Biological Conservation* 43:63-76.
- Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. *Nature* 405:243-253.
- Molnar, J., M. Marvier, and P. Kareiva. 2004. The sum is greater than the parts. *Conservation Biology* 18:1670-1671.

- Moore, J. W., and D. P. Moore. 1987. *The Army Corps of Engineers and the evolution of federal flood plain management policy*. Program on Environment and Behavior, Special Publication No. 20. Boulder, CO: Institute of Behavioral Science, University of Colorado.
- Muschett, F. D. 1997. An integrated approach to sustainable development. In *Principles of Sustainable Development*, ed. F. D. Muschett, 1-46.. Delray, FL: St. Lucie Press.
- NatureServe. 2009. NatureServe Explorer: An online encyclopedia of life. <http://www.natureserve.org/explorer/index.htm>
- Newburg, D., S. Reed, P. Berck, and A. Merenlender. 2005. Economics and land-use change in prioritizing private land conservation. *Conservation Biology* 19: 1411-1420.
- Niven, P. R. 2003. *Balanced scorecard: Step-by-step for government and nonprofit agencies*. Wiley, Hoboken, NJ
- NOAA Panel on Contingent Valuation. 1993. Natural resource damage assessment under the Oil Pollution Act of 1990. *Federal Register* 58(10) 4601-4614.
- Norton, B. G.(Editor). 1986. *The Preservation of Species: The Value of Biological Diversity*. Princeton NJ: Princeton University Press.
- Noss R. F., and A. Y. Cooperrider 1994. *Saving Nature's Legacy: Protecting and Restoring Biodiversity*. Washington DC: Island Press.
- Noss, R. F., E. T. Laroe III, and J. M. Scott. 1995. *Endangered ecosystems of the United States: A preliminary assessment of loss and degradation*. Biological Report 28. Washington, DC: USDI National Biological Service.
- Noss, R. F., M. A. O'Connel and D. D. Murphy. 1997. *The Science of Conservation Planning: Habitat Conservation under the Endangered Species Act*. Washington, DC: Island Press.
- Noss, R. F., E. Dinerstein, B. Gilbert, M. Gilpin, B. J Miller, J. Terbourgh, and S. Trombulak. 1999. Core areas: where nature begins. In *Continental Conservation: Scientific Foundations of Regional Reserve Networks* ed. J. Terorgh and M. Soule, 99-128. . Washington, DC: Island Press.
- National Research Council (NRC) 1993. *Setting Priorities of Land Acquisition*. Washington, DC: National Academy Press.
- _____. 1995. *Science and the Endangered Species Act*. Washington, DC: National Academy Press.
- _____. 1996. *Upstream: Salmon and Society in the Pacific Northwest*. Washington, DC: National Academy Press.
- _____. 1999a. *Our Common Journey: A Transition toward Sustainability*. Washington, DC: National Academy Press.

- _____. 1999b. Downstream: Adaptive Management of Glen Canyon Dam and the Colorado River Ecosystem. Washington, DC: National Academy Press.
- _____. 2000. *Ecological Indicators for the Nation*. Washington, DC: National Academy Press.
- _____. 2005. Valuing Ecosystem Services: Toward Better Environmental Decision Making. Washington, DC: National Academy Press.
- Office of Management and Budget (OMB). 1992. *Guidelines and Discount Rates for Benefit-Cost Analysis of Federal Programs*. Circular A-94. Washington DC: OMB. <http://www.whitehouse.gov/omb/circulars/a094/a094.html>.
- _____. 2003. *Regulatory Analysis*. Circular A-4. Washington, DC: OMB. <http://www.whitehouse.gov/omb/circulars/a004/a-4.pdf>.
- _____. 2006a. Guidance for completing 2006 PARTs. Program Assessment Rating Tool Guidance No. 2006-02. Washington, DC: OMB.
- _____. 2006b. Guidance for improving the consistency of 2006 PART assessments. Program Assessment Rating Tool Guidance No. 2006-04. Washington, DC: OMB.
- _____. 2007a. Historical tables, fiscal year 2007 budget of the United States government. Washington, DC: US Government Printing Office.
- _____. 2007b. *Fiscal year 2007, budget of the US government*. Washington, DC: US Government Printing Office.
- O'Neil, J., S. Whitaker, and M. Grodowitz. 2001. *Ecosystem Management and Restoration System (EMRIS)*. Vicksburg, VA: US Army Engineer Research and Development Center.
- Orr, D. 2003. The constitution of nature. *Conservation Biology* 17: 1478-1484.
- Orians, G. H. 1993. Endangered at what level? *Ecological Applications* 3:206-208.
- Pagiola, S., K. von Ritter, and J. T. Bishop. 2004. *Assessing the economic value of ecosystem conservation*. Environmental Development Paper No. 101. Washington, DC: World Bank.
- Pearsall, S. 1984. In absentia benefits of natural preserves: A review. *Environmental Conservation* 11(1): 3-10.
- Perlman, D. L., and G. Adelson. 1997. *Biodiversity: Exploring Values and Priorities in Conservation*. Malden, MA: Blackwell Science.
- Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross, and R. M. Hughes. 1989. *Rapid bioassessment protocols for use in streams and rivers: Benthic macroinvertebrates and fish*. EPA/440/ 41-89-001. Washington DC: US Environmental Protection Agency.

- Polasky, S. J., D. Camm, and B. Garter-Yonts. 2001. Selecting biological reserves cost-effectively: An application to terrestrial vertebrate conservation in Oregon. *Land Economics* 77:68-78.
- Portnoy, P. R. 1990. Overall assessment and future directions. In Public policies for environmental protection. Resources for the Future, ed. P. R. Portnoy, 275 -290. Baltimore, MD: Johns Hopkins University Press.
- Portnoy, P. R. 1994. The contingent valuation debate: Why economists should care. *The Journal of Economic Perspectives* 8(4):3-18.
- Pressey, R. L, and A. O. Nicholls. 1989. Application of a numerical algorithm to the selection of reserves in semi-arid New South Wales. *Biological Conservation* 50:263-278.
- Pressey, R. L., C. J. Humphries, C. R. Margules, R. I. Vane-Wright, and P. H. Williams. 1993. Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology and Evolution* 8:124-128.
- Pressey, R. L. 2004. Conservation planning and biodiversity: Assembling the best data for the job. *Conservation Biology* 18:1677-1681
- Pressey, R. L., M. Cabeza, M.E. Watts, R. M. Cowling and K. A. Wilson. 2007. Conservation planning in a changing world. *Trends in Ecology and Evolution* 22 (11): 583-592.
- Randall, A. 1991. Total and nonuse values. In *Measuring the Demand for Environmental Quality*, ed. J. B. Braden and D. D. Kolstad, Chapter 10 . Amsterdam, Netherlands: North Holland.
- Randall, A., and M. C. Farmer. 1995. Benefits, costs, and the safe minimum standard of conservation. In *The Handbook of Environmental Economics*, ed. D. W. Bromley. Oxford, UK: Blackwell.
- Reid, W. V., and K. R. Miller. 1989. *Keeping options alive: The scientific basis for conserving biodiversity*. Washington, DC: World Resources Institute.
- Rohlf, D. J. 1989. *The Endangered Species Act: A guide to its protections and implementation*. Stanford, CA: Stanford Environmental Law Society.
- Rosensweig, M. L. 1995. *Species Diversity in Space and Time*. New York, NY: Cambridge University Press.
- Sagoff, M. 1997. Can we put a price on nature's services? *Philosophy and Public Policy* 17(3):7-12.
- Sale, K. 1993. *The Green Revolution: The American Environmental Movement 1962-1992*. New York, NY: Hill and Wang.
- Schaffer, M. L., and B. L. Stein. 2000. Safeguarding our precious heritage. In *Precious Heritage: The Status of Biodiversity in the United States*, ed. B. A. Stein, L. S. Kutner, and J. S. Adams, 301-322. Oxford, UK: Oxford University Press.

- Scott, J. M., D. D. Goble, L. K. Svancara, and A. Pidgorna. 2006. By the numbers. In *The Endangered Species Act at Thirty: Renewing the Conservation Promise*, Volume 1, ed. D. D. Goble, J. M. Scott, and F. W. Davis, 16-35.. Washington, DC: Island Press.
- Shabman, L. 1997. Making benefit estimation useful: Lessons from flood control experience. *Water Resources Update* 109: 19-24
- Smith, P. G. R. and J. B. Theberge. 1986. A review of criteria for evaluating natural areas. *Environmental Management* 10: 715-734.
- Smith, R. D., A. Ammann, C. Bartoldus, and M. M. Brinson. 1995. *An approach for assessing wetland functions using hydrogeomorphic classification, reference wetlands, and functional indices*. Wetlands Research Program Technical Report WRP-DE-9. Vicksburg, MS: US Army Environmental Laboratory. .
- Soule, M. E., and J. Terborgh. 1999. Conserving nature at regional and continental scales—a scientific program for North America. *Bioscience* 49:808-818.
- Stakhiv, E., R. Cole, P. Scodari, and L. Martin. 2003. *Improving environmental benefits analysis in ecosystem restoration planning*. IWR Report 03-PS-3. Alexandria, VA: Policy Studies Program, U S. Army Institute for Water Resources.
- Stein, B. A., L. S. Kutner, and J. S. Adams. 2000. *Precious Heritage: The Status of Biodiversity in the United States*. New York, NY: Oxford University Press.
- Suckling, K., and M. Taylor. 2006. Critical habitat and recovery. In *The Endangered Species Act at Thirty: Renewing the Conservation Promise*, Volume 1, ed. D. D. Goble, J. M. Scott, and F. W. Davis. 75-89.. Washington, DC: Island Press.
- Taylor, P. W. 1986. *Respect for Nature: A Theory of Environmental Ethics*. Princeton, NJ: Princeton University Press.
- The H. John Heinz III Center for Science, Economics and the Environment. 2008. *The State of the Nation's ecosystems: Measuring the lands, waters, and living resources of the United States*. Washington D C: Island Press.
- The Nature Conservancy (TNC). 1997. Designing a geography of hope: Guidelines for ecoregion-based conservation in The Nature Conservancy. Arlington, VA: The Nature Conservancy.
- Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. *Landscape Ecology in Theory and Practice: Pattern and Process*. New York, NY: Springer-Verlag.
- US Army Corps of Engineers (USACE) . 2002. Final estuary habitat restoration strategy prepared by the Estuary Habitat Restoration Council. Notice. *Federal Register*. Vol. 67, No. 232: 71942-71949.
- Usher, M. B. 1986. Wildlife conservation evaluation: attributes, criteria, and values. In *Wildlife Conservation Evaluation*, ed. M. B. Usher, 3-44. London, UK: Chapman and Hall
- Varner, G. E. 1999. *In Nature's Interests? Interests, Animal Rights, and Environmental Ethics*. New York, NY: Oxford University Press.

- Vaughn, I. P. and S. J. Ormerod. 2003. Improving the quality of distribution models for conservation by addressing shortcomings in the field collection of training data. *Conservation Biology* 17: 1601-1611.
- Wilson, K. A., M. F. McBride, M. Bode and H. E. Passingham. 2006. Prioritizing global conservation efforts. *Nature* 440: 337-340.
- WCED (*World Commission on Environment and Development*). 1987. *Our Common Future*. New York, NY: Oxford University Press.
- WRC (US Water Resources Council). 1973. Principles and standards for planning water and related land resources. Notice of Establishment, *Federal Register* Vol. 38, No. 174: 24779-24868.
- WRC (US Water Resources Council) 1983. *Economic and environmental principles and guidelines for water and related resources implementation studies*. Washington, DC: US Army Water Resources Council.. http://www.usace.army.mil/cw/cecw-cp/library/Principles_Guidelines.pdf
- Yaffee, S. L. 1982. *Prohibitive Policy: Implementing the Federal Endangered Species Act*. Cambridge, MA: MIT Press.

REPORT DOCUMENTATION PAGE				Form Approved OMB No. 0704-0188	
Public reporting burden for this collection of information is estimated to average 1 hour per response, including the time for reviewing instructions, searching existing data sources, gathering and maintaining the data needed, and completing and reviewing this collection of information. Send comments regarding this burden estimate or any other aspect of this collection of information, including suggestions for reducing this burden to Department of Defense, Washington Headquarters Services, Directorate for Information Operations and Reports (0704-0188), 1215 Jefferson Davis Highway, Suite 1204, Arlington, VA 22202-4302. Respondents should be aware that notwithstanding any other provision of law, no person shall be subject to any penalty for failing to comply with a collection of information if it does not display a currently valid OMB control number. PLEASE DO NOT RETURN YOUR FORM TO THE ABOVE ADDRESS.					
1. REPORT DATE (DD-MM-YYYY) May 2014		2. REPORT TYPE Final report		3. DATES COVERED (From - To)	
4. TITLE AND SUBTITLE Measuring Environmental Value in Nonmonetary Terms: A Review of Common Practices and Elements				5a. CONTRACT NUMBER	
				5b. GRANT NUMBER	
				5c. PROGRAM ELEMENT NUMBER	
6. AUTHOR(S) Richard A. Cole				5d. PROJECT NUMBER	
				5e. TASK NUMBER	
				5f. WORK UNIT NUMBER	
7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES) Institute for Water Resources, United States Army Corps of Engineers 7701 Telegraph Road, Casey Building Alexandria VA 22315-3868				8. PERFORMING ORGANIZATION REPORT NUMBER ERDC/EL CR-14-1	
9. SPONSORING / MONITORING AGENCY NAME(S) AND ADDRESS(ES) Headquarters, US Army Corps of Engineers Washington, DC 20314-1000				10. SPONSOR/MONITOR'S ACRONYM(S)	
				11. SPONSOR/MONITOR'S REPORT NUMBER(S)	
12. DISTRIBUTION / AVAILABILITY STATEMENT Approved for public release; distribution is unlimited.					
13. SUPPLEMENTARY NOTES					
14. ABSTRACT This review was undertaken to address concerns raised by the US Army Corps of Engineers (Corps) regarding the value of projects authorized to improve environmental quality. Value gained from present resource use can typically be measured in monetary terms. More controversial -- and prohibited by Corps policy -- is the monetary measurement of the nonuse value gained by deferring present-day use in favor of leaving a heritage for future generations. Environmental value is commonly indicated by the objectives of government legislation and by non-government mission statements and bylaws. The value added by objective achievement is indicated by many different, incomparable metrics, which often do not differentiate use value from nonuse value. The Corps is an exception because Federal policy requires water resources agencies to quantify benefits and costs in monetary terms when acceptable and in non-monetary terms when monetary measurement is not acceptable. This includes heritage value recognized as important by the Corps in key environmental legislation, and by certain conservation NGOs. Key elements of natural heritage value include resource security from extinction, resource distinctiveness, risk of investment failure, and costs. These elements may provide a basis for comprehensively indicating the value added by ecosystem restoration done by the Corps.					
15. SUBJECT TERMS Ecosystem Management and Restoration Research Program (EMRRP) Ecosystem restoration Environmental value Environmental quality (EQ) Natural heritage value					
16. SECURITY CLASSIFICATION OF:			17. LIMITATION OF ABSTRACT	18. NUMBER OF PAGES	19a. NAME OF RESPONSIBLE PERSON
a. REPORT UNCLASSIFIED	b. ABSTRACT UNCLASSIFIED	c. THIS PAGE UNCLASSIFIED			19b. TELEPHONE NUMBER (include area code)
				79	