Valuing Ecosystem Services:

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VALUING ECOSYSTEM SERVICES:

AN ECOLOGICAL ECONOMIC APPROACH

A Dissertation Presented

by

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to

The Faculty of the Graduate College

of

The University of Vermont

In Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy
Specializing in Natural Resources

October, 2007

Accepted by the Faculty of the Graduate College, The University of Vermont, in partial fulfillment of the requirements for the degree of Doctor of Philosophy, specializing in Natural Resources.

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ABSTRACT

Ecosystem services are the benefits people obtain from ecosystems. Ecosystem service valuation (ESV) is the process of assessing the contributions of ecosystem services to human well-being. Its goal is to express the effects of changes in ecosystem services in terms of trade-offs against other things that also support human welfare. Ecologists tend to use biophysical-based methods while economists have developed preference-based tools for ESV. In this dissertation I attempt to bridge these two worlds by writing *six* papers using methods and insights from both disciplines.

In *paper 1,* my coauthors and I (thereafter "we") reviewed (1) what has been done in ESV research in the last 45 years; (2) how it has been used in ecosystem management; and (3) prospects for the future. One conclusion is that researchers and practitioners will have to transcend disciplinary boundaries and synthesize methodologies and tools from various disciplines in order to meet the challenge of ecosystem service valuation and management.

Ninety-four peer-reviewed environmental economic studies were used to value ecosystem services in the State of New Jersey in *paper 2*. We translated each benefit estimate into 2004 US dollars per acre per year, computed the average value for a given eco-service for a given ecosystem type, and multiplied the average by the total statewide acreage for that ecosystem. The total value of these ecosystem services was estimated as $11.6 billion/year and we believe that this result is conservative. This aggregate value of New Jersey's ecosystem services is a useful, albeit imperfect, basis for assessing and comparing these services with conventional economic goods and services.

In *paper 3* we present a conceptual framework for non-market valuation of ecosystem services provided by coastal and marine systems and review the peer-reviewed literature in this area. Next we selected a subset of this literature and conducted the first meta-analysis of the ecosystem service values provided by the costal and nearshore marine systems in *paper 4*. Using regression we found that over 75% of the variation in willingness to pay (WTP) for coastal ecosystem services could be explained. Our meta-regression models also predicted out-of-sample WTPs and showed that the overall average transfer error was 24%, with 40% of the sample having transfer errors of 10% or less, and only 2.5% of predictions having transfer errors of over 100%.

In the final two papers our focus is on the linkage between biodiversity and ecosystem function (BEF) which connects ecosystems with human welfare. In *paper 5* we first present an overview of the main concepts and findings from a decade of the BEF literature. After a discussion on how agrobiodiversity relates to stability and resilience in agricultural systems at both the species and the landscape scales, we conclude with observations on the research needs in assessing the BEF relationship and the implications for agrobiodiversity ESV research. Finally, in *paper 6*, by using multiple regression analysis at the site and ecoregion scales in North America, we estimated relationships between biodiversity (using plant species richness as a proxy) and Net Primary Production (NPP, as a proxy for ecosystem services). We tentatively conclude that a 1% change in biodiversity in the high temperature range (which includes most of the world's biodiversity) corresponds to approximately a 1/2% change in the value estimate of ecosystem services.

DEDICATION

This dissertation is dedicated to my grandmother, who could hardly read, but taught me the meaning of love and dedication.

ACKNOWLEDGEMENTS

I would like to thank my doctoral committee members, Robert Costanza, Marta Ceroni, Austin Troy, and Matthew Wilson for their generous time and commitment. Throughout my doctoral work they encouraged me to develop independent thinking and research skills. They continually stimulated my analytical thinking and greatly assisted me with scientific writing.

I am also very grateful for having an extended Gund family and wish to thank all the members for their support and encouragement. I have never felt foreign in a foreign country ever since the first day in the Institute.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Page

Paper 4

Paper 5

Paper 6

Valuing ecosystem services: theory, practice and the need for a trans-disciplinary synthesis*

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* An early version of this paper was published as Appendix A in Costanza et al. (2007).

ABSTRACT

The concept of ecosystem services has shifted our paradigm regarding how nature matters to human societies. Instead of something we have to sacrifice our wellbeing to preserve, we now think of the natural environment as natural capital, one of society's important assets. Ecosystem services valuation (ESV) is the process of evaluating the effects of changes in ecosystem services against other things that also support human welfare. It provides a tool that enhances the ability of decision-makers to evaluate trade-offs between alternative ecosystem management regimes. This review covers: (1) what has been done in ESV research in the last 45 years; (2) how it has been used in ecosystem management; and (3) prospects for the future. One conclusion is that researchers and practitioners will have to transcend disciplinary boundaries and synthesize methodologies and tools from various disciplines in order to meet the challenge of ecosystem service valuation and management.

KEYWORDS: Ecosystem service valuation; Trans-disciplinary; Environmental decision-making

Ecosystem services are the benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as regulation of floods, drought, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual and other nonmaterial benefits (Costanza et al. 1997, Daily 1997, de Groot et al. 2002).

Ecosystem services are becoming scarcer. On the supply side, ecosystems are experiencing serious degradation in regard to their capability of providing services. At the same time, the demand for ecosystem services is increasing rapidly as populations and standards of living increase (Millennium Ecosystem Assessment 2005).

Value, Valuation and Social Goals

In discussing values, we first need to clarify some underlying concepts and definitions. The following definitions are based on Farber et al. (2002).

"Value systems" refer to intrapsychic constellations of norms and precepts that guide human judgment and action. They refer to the normative and moral frameworks people use to assign importance and necessity to their beliefs and actions. Because "value systems" frame how people assign importance to things and activities, they also imply internal objectives. Value systems are thus internal to individuals, but are the result of complex patterns of acculturation and may be externally manipulated through, for example, advertising.

"Value" refers to the contribution of an object or action to specific goals, objectives or conditions (Costanza 2000). The value of an object or action may be tightly coupled with an individual's value system because the latter determines the relative

importance to the individual of an action or object relative to other actions or objects within the perceived world. But people's perceptions are limited, they do not have perfect information, and they have limited capacity to process the information they do have. An object or activity may therefore contribute to meeting an individual's goals without the individual being fully (or even vaguely) aware of the connection. The value of an object or action therefore needs to be assessed both from the "subjective" point of view of individuals and their internal value systems, and also from the "objective" point of view of what we may know from other sources about the connection.

"Valuation" is then the process of assessing the contribution of a particular object or action to meeting a particular goal, whether or not that contribution is fully perceived by the individual. A baseball player is valuable to the extent he contributes to the goal of the team's winning. In evolutionary biology, a gene is valuable to the extent it contributes to the survival of the individuals possessing it and their progeny. In conventional economics, a commodity is valuable to the extent it contributes to the goal of individual welfare as assessed by willingness to pay. The point is that one cannot state a value without stating the goal being served (Costanza 2000).

"Intrinsic value" refers more to the goal or basis for valuation itself and the protection of the "rights" of these goals to exist. For example, if one says that nature has "intrinsic value" one is really claiming that protecting nature is an important goal in itself. "Values" (as defined above) are based on the contribution that something makes toward achieving goals (directly or indirectly). One could thus talk about the value of an object or action in terms of its contribution to the goal of preserving nature, but not about the "intrinsic value" of nature. So "intrinsic value" is a confusing term. Because intrinsic

value is a goal, one cannot estimate or measure the intrinsic value of something and compare it with the intrinsic value of something else. One should therefore more accurately refer to the "intrinsic rights" of nature to qualify as a goal against which to assess value, in addition to the more conventional economic goals.

ESV is thus the process of assessing the contribution of ecosystem services to meeting a particular goal or goals. Traditionally, this goal is efficient allocation, that is, to allocate scarce ecosystem services among competing uses such as development and conservation. But other goals, and thus other values, are possible.

There are at least three broad goals that have been identified as important to managing economic systems within the context of the planet's ecological life support system (Daly 1992):

1) assessing and insuring that the scale or magnitude of human activities within the biosphere are ecologically sustainable;
2) distributing resources and property rights fairly, both within the current generation of humans and between this and future generations, and also between humans and other species; and
3) efficiently allocating resources as constrained and defined by 1 and 2 above, and including both market and non-market resources, especially ecosystem services.

Because of these multiple goals, one must do valuation from multiple perspectives, using multiple methods (including both subjective and objective), against multiple goals (Costanza 2000). Furthermore, it is important to recognize that the three goals are not ‘‘either–or’’ alternatives. Whereas they are in some sense independent multiple criteria

(Arrow and Raynaud 1986) which must all be satisfied in an integrated fashion to allow human life to continue in a desirable way.

However, basing valuation on current individual preferences and utility maximization alone does not necessarily lead to ecological sustainability or social fairness (Bishop 1993), or to economic efficiency for that matter, given the severe market imperfections involved. ESV provides a tool that enhances the ability of decision-makers to evaluate trade-offs between alternative ecosystem management regimes in order to meet a set of goals, namely, sustainable scale, fair distribution, and efficient allocation (Costanza and Folke 1997). Different goals may become a source of conflict during policy-making debates over management of ecosystem services. How are such conflicts to be resolved? ESV provides one approach to at least better inform these discussions.

Framework for ESV

Figure 1 shows one integrated framework developed for ESV (from de Groot et al. 2002). It shows how ecosystem goods and services form a pivotal link between human and ecological systems. Ecosystem structures and processes are influenced by biophysical drivers (i.e., tectonic pressures, global weather patterns, and solar energy) which in turn create the necessary conditions for providing the ecosystem goods and services that support human welfare. Through laws, land use management and policy decisions, individuals and social groups make tradeoffs. In turn, these land use decisions directly modify the ecological structures and processes by engineering and construction activities and/or indirectly by modifying the physical, biological and chemical structures and processes of the landscape.

[Insert Figure 1]

Methodology for ESV

Because there are no markets for most ecosystem services, a spectrum of valuation techniques have been developed to value them (Freeman 2003, Champ et al. 2003, US National Research Council, 2005). These include both nonmonetizing valuation methods as well as conventional economic techniques based on a common metric, normally a monetary metric (Box 1). The use of a dollar metric assumes individuals are willing to trade the ecosystem service being valued for other goods or services represented by the metric. The purpose of economic valuation is to allow measurement of the costs or benefits associated with changes in ecosystem services, using a common metric.

[Insert Box 1]

The principle distinction among these economic valuation methods is based on the data source, that is, whether they come from observations of people's behavior in the real-world (i.e. *revealed-preference approaches*) or from people's responses to hypothetical questions (*state-reference approaches*) such as "How much would you be willing to pay for…?" or "What would you do if…?".

When an ecosystem service is difficult to value using any of the above methods, researchers (mainly ecologists) have resorted to using the method of *replacement/avoided cost*. However economists believe these cost-based approaches should be used with great caution if at all (Shabman and Batie 1978, Bockstael 2000, US National Research Council 2005). This is because any value estimates derived from such approaches should be on the cost side of the benefit-cost ledger, not counted as a benefit, and the conditions under which these cost estimates can serve as a last resort proxy are often too rigid to be met.

Conducting original valuation research is expensive and time-consuming. As a “second-best” strategy, in the last few decades *benefit transfer* has been applied as decision makers seek a timely and cost-effective way to value ecosystem services (Wilson and Hoehn 2006). It involves obtaining an estimate for the value of ecosystem services through the analysis of a single study or group of studies that have been previously carried out to value “similar” goods or services in “similar” locations. The transfer itself refers to the application of derived values and other information from the original ‘study site’ to a ‘policy site’ which can vary across geographic space and/or time (Brookshire and Neill 1992, Desvouges et al. 1992).

The ability to transfer values from one context to another is service-specific. Some ecosystem services, such as carbon sequestration, may be provided at a scale in which benefits are easily transferable. On the contrary other local-scale services may have limited transferability, such as flood control values. Table 1 provides guidance for transferring service values from one context to another (Farber et al. 2006).

Similarly Table 1 also illustrates some valuation tools are more appropriate for an ecosystem service than for others. For example, travel Cost (TC) is primarily used for estimating recreation values while Hedonic Pricing (HP) for estimating property values associated with aesthetic qualities of natural ecosystems. Contingent Valuation (CV) and Conjoint Analysis (CA) are the only methods to measure non-use values like existence value of wildlife[1]. Finally, nonmonetizing methods do not require valuation results expressed in a single monetary unit. For instance, group valuation (GV) is a more recent addition to the valuation literature and addresses the need to measure social values directly in a group context (Wilson and Howarth 2002, Howarth and Wilson 2006). In many applications, the full suite of ecosystem valuation techniques will be required to account for total value of goods and services provided by a natural landscape.

[Insert Table 1]

History of ESV Research

This section provides a historical perspective on ESV research. For the purpose of this paper the story opens with the emergence of environmentalism in the 1960s. However, this is not to say that the foundations of ESV were not present prior to this. For instance, Hotelling's (1949) discussion of the value of parks implied by travel costs signaled the start of the travel cost valuation era. Similarly suggestions by Ciriacy-Wantrup (1947) in the late 1940s led to the use of stated preference techniques such as contingent valuation.

Our approach to the history of advances in ESV will not be a method by method literature review[2]. Rather, we focus on how people faced the challenge presented by the transdisciplinary nature of ESV research. In the 1960s, for instance, there was relatively little work that transcended disciplinary boundaries on ecosystem services. In later years

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[1]The concept of economic value is much more inclusive than people often thought. For instance, many of what are typically considered non-economic values are in fact to some degree captured by "*existence value*".

[2] Several reviews of the published ESV literature have been developed elsewhere. These review, including Smith (1993, 2000), Carson (2000), Cropper (2000), Freeman (2003), Champ et al. (2003) provided a much more detailed examination of ESV methods.

this situation has gradually improved. Truly *transdisciplinary* approaches are required for ESV in which practitioners accept that disciplinary boundaries are academic constructs largely irrelevant outside of the university, and allow the problem being studied to determine the appropriate set of tools, rather than vice versa.

We frequently see ESV research in which teams of researchers trained in different disciplines separately tackle a single problem and then strive to combine their results. This is known as *multidisciplinary* research, but the result is much like the blind men who examine an elephant, each describing the elephant according to the single body part they touch. The difference is that the blind men can readily pool their information, while different academic disciplines lack even a common language with which their practitioners can communicate (e.g. Bingham and others 1995). *Interdisciplinary* research, in which researchers from different disciplines work together from the start to jointly tackle a problem and reduce the language barrier as they go, is a step in the right direction toward the transdisciplinary path.

For convenience, we arbitrarily divide the last 45 years (1960 to present) into four periods. Influential contributions during each period are marked as milestones in Figure 2. The chart is meant to be illustrative, not comprehensive, as space prohibits showing all important contributions and milestones.

[Insert Figure 2]

<u>1960s—Common challenge, separate answers</u>

The 1960s are remembered as the decade of early environmentalism. Main social events include publication of Rachel Carson's *Silent Spring* in 1962, passage of the 1970

Clean Air Act, and formation of the U.S. Environmental Protection Agency in that same year.

In response to increasing public interest in environmental problems (mainly pollution and dramatic population increase at the time[3]), economists began rethinking the role of the environment in their production models and identified new types of surplus for inclusion in their welfare measure (Crocker 1999).

Economist Kenneth Boulding compared the "cowboy economy" model which views the environment as a limitless resource with the "spaceship economy" view of the environment's essential limits (Boulding 1966). His work included recognition of the ecosystem service of waste assimilation to the production model, where before ecosystems had mainly been regarded as a source of provisioning services.

Consideration of cultural services in an economic analysis began with Krutilla's (1967) seminal observation that many people value natural wonders simply for their existence. Krutilla argued that these people obtain utility through vicarious enjoyment of natural areas and, as a result, had a positive WTP for the government to exercise good stewardship of the land.

In addition to *existence value*, other types of value were also considered. These include *option value*[4], or the value of avoiding commitments that are costly to reverse (Weisbrod 1964). There is also *quasi-option value*, or the value of maintaining opportunities to learn about the costs and benefits of avoiding possibly irreversible future states (Arrow and Fisher 1974).

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[3] The population issue was brought to the forefront by Paul Ehrlich in the provocative book *the Population Bomb* (1968). As a biologist, he had an inclination to perceive human beings as a species and deeply questioned the sufficiency of food production when human population increases dramatically.

[4] Option value is not a component of Total Economic Value (TEVs). It is the concept of TEV when uncertainty is present and includes all use and nonuse values.

In most cases, WTPs for these newly-recognized values could not be derived via market transactions because most of the ecosystem services in question are not traded in actual markets. Thus, new valuation methods were also proposed, including *travel cost* (Clawson 1959), *contingent valuation* (Davis 1963), and *hedonic pricing* (Ridker and Henning 1967).

In the meantime, ecologists also proposed their own valuation methods. For example, "*energy analysis*" is based on thermodynamic principles where solar energy is considered to be the only primary input to the global ecosystem (Odum 1967). This biophysical method differs from WTP-based ones in that it does not assume that value is determined by individual preferences, but rather attempts a more "objective" assessment of ecosystem contributions to human welfare.

1970s—breaking the disciplinary boundary

The existence of "limits to growth" was the main message in the environmental literature during the 1970s (Meadows et al. 1972). The Arab oil embargo in 1973 emphasized this message.

"*Steady-state economics*" as an answer to the growth limit was proposed by economist Herman Daly (1977), who emphasized that the economy is only a sub-system of the finite global ecosystem. Thus the economy cannot grow forever and ultimately a sustainable steady state is desired. Daly was inspired by his mentor in graduate school, Nicholas Georgescu-Roegen. In *The Entropy Law and the Economic Process*, Georgescu-Roggen elaborates extensively on the implications of the entropy law for economic processes and how economic theory could be grounded in biophysical reality (Georgescu-Roegen 1971).

Georgescu-Roegen was not the only scientist to break the disciplinary boundary in the 1970s. Ecologist H.T. Odum published his influential book *Environment, Power, and Society* in 1971, where he summarized his insights from studying the energetics of ecological systems and applying them to social issues (Odum 1971).

Along with these early efforts, a rather heated debate between ecologists and economists also highlighted their differences regarding concepts of value. The economists of the day objected strenuously to the energetic approach. They contended that value and price were determined solely by people's ''willingness to pay'' and not by the amount of energy required to produce a service. H. T. Odum and his brother E. P. Odum and economists Leonard Shabman and Sandra Batie engaged in a point–counterpoint discussion of this difference in the pages of the *Coastal Zone Management Journal* (Shabman and Batie 1978, EP Odum 1979, HT Odum 1979).

Though unrealized at the time, a new method called the *production function* approach became one way to bring together the views of ecologists and economists. This method is used to estimate the economic value of ecosystem services that contribute to the production of marketed goods. It is applied in cases where ecosystem services are used, along with other inputs, to produce a market good (cf. McConnell and Brockstael 2006 for a review and Barbier 2007 for examples in valuing habitat and storm protection service).

Early contributions in the area include works from Anderson (1976), Schmalensee (1976), and Just and Hueth (1979). Just and his colleagues (1982) provided a rigorous analysis of how to measure changes in welfare due to price distortions in factor and product markets. These models provide a basis for analyzing the effects of productivity-induced changes in product and factor prices.

The field of environmental and resource economics grew rapidly from the beginning of the 1970s. The field became institutionalized in 1974 with the establishment of the *Journal of Environmental Economics and Management* (JEEM). The objects of analysis for natural resource economists have typically been such resources as forests, ore deposits, and fish species that provided provisioning services to the economy. In the meantime, the environment has been viewed as the *medium* through which the externalities associated with air, noise, and water pollution have flowed, as well as the source of amenities. However, in later years this distinction between natural resources and the environment has been challenged as artificial and thus no longer meaningful or useful (Freeman 2003).

1980s—moving beyond multidisciplinary ESV research

In the 1980s, two government regulations created a tremendous demand for valuation research. The first was the 1980 *Comprehensive, Environmental Responses, Compensation and Liability Act* (CERCLA), commonly known as *Superfund*, which established liability for damages to natural resources from toxic releases. In promulgating its rules for such Natural Resource Damage Assessments (NRDA), the US Department of Interior interpreted these damages and the required compensation within a welfare-economics paradigm, measuring damages as lost consumer surplus. The regulations also describe protocols that are based on various economic valuation methods (Hanemann 1992).

The role of ecosystem valuation increased in importance in the United States with President Reagan's Executive Order 12911, issued in 1981, requiring that all new major regulations be subject to a Cost Benefit Analysis (CBA) (Smith 1984).

As shown in Figure 3, the 1980s witnessed dramatic increases in the number of publications, including peer-reviewed papers, book chapters, governmental reports, and theses, on the topic of ecosystem valuation[5]. These results are based on a search of the Environmental Valuation Reference Inventory ™ (EVRI™), the largest valuation database. The search was conducted for four general types of entities relevant to ecosystem services including ecological functions, extractive uses, non-extractive uses, and passive uses. We excluded valuation publications on human health and the built environment from EVRI™ because they are not relevant to ESV.

[Insert Figure 3]

The 1989 *Exxon Valdez* oil spill was the first case where non-use value estimated by contingent valuation was considered in a quantitative assessment of damages. In March of that year, the *Exxon Valdez* accidentally spilled eleven million gallons of oil in Alaska's pristine Prince William Sound. Four months later, the District of Columbia Circuit of the US Court of Appeals held that non-use value should be part of the economic damages due to releases of oil or hazardous substances that injure natural resources. Moreover, the decision found that CV was a reliable method for undertaking such estimates. Prior to the spill, CV was not a well developed area of research. After the widely publicized oil spill, the attention given to the conceptual underpinnings and estimation techniques for non-use value increased rather abruptly (Carson et al. 2003). In

[5] The drop of the number of publications in some recent years is probably due to artificial effect, i.e. EVRI™ has not included all the publications. According to a similar analysis by Adamowicz (2004), the amount of peer-reviewed literature in environmental valuation has increased over time.

the same year, two leading researchers published their state-of-the-art work on CV (Mitchell and Carson 1989).

At the same time, ecologists began to compare their results based on energy analysis to conventionally derived economic values. For example, Costanza (1980) and Costanza and Herendeen (1984) used an 87-sector input-output model of the US economy for 1963, 1967, and 1973, modified to include households and governments as endogenous sectors, to investigate the relationship between direct and indirect energy consumption (*embodied energy*[6]) and the dollar value of output by sector. They found that the dollar value of sector output was highly correlated with embodied energy, though not with direct energy consumption or with embodied energy calculated excluding labor and government energy costs.

Differences of opinion between ecologists and economists still existed in the 1980s in terms of the relationship between energy inputs, prices, and values (Ropke 2004). But the decade also witnessed the first paper co-authored by an ecologist and an economist on ecosystem valuation (Farber and Costanza 1987). Though the idea of the paper was simply to compare the results from two separate studies using different methods, the paper also represented the first instance of an ecologist and economist overcoming their disciplinary differences and working together.

The term *Ecosystem Services*, first appeared in Ehrlich and Ehrlich's work (1981). The concept of ecosystem services represents an attempt to build a common language for discussing linked ecological and economic systems. Using "ecosystem services" and

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[6] The energy embodied in a good or service is defined as the total direct energy used in the production process plus all the indirect energy used in all the upstream production processes used to produce the other inputs to the process. For example, auto manufacturing uses energy directly, but it also uses energy indirectly to produce the steel, rubber, plastic, labor, and other inputs needed to produce the car.

"environmental services" as key words, a search in the ISI Web of Knowledge show the total number of papers published and the number of disciplinary categories in which they occur over time (Figure 4). For example, the curves indicate that by the year 2006, more than to 200 papers per year were being published on ecosystem services - in about 50 subdisciplines. The two exponential curves show the increasing use of the term over time and the fact that it has been embraced quickly by many different disciplines, including those which appear at first glance to be not so relevant, such as computer science, pharmacy, business, law and demography.

[Insert Figure 4]

The concept of ecosystem services and the related concept of "*natural capital*[7]" have enhanced our understanding of how the natural environment matters to human societies. It is now believed that the natural environment and the ecosystems within are natural capital, along with the physical, human, and social capitals, and these four all together comprise society's important assets.

<u>1990s ~ present: Moving toward trans-disciplinary ESV research</u>

Not only attention but also controversy was drawn to the CV approach after its application to the *Exxon Valdez* case, when it became known that a major component of the legal claims for damages was likely to be based on CV estimates of lost nonuse or existence value. The concerns about the reliability of the CV approach led the National Oceanic and Atmospheric Administration (NOAA) to convene a panel of eminent experts

[7] Natural capital is defined as the stock of ecosystem structure that produces the flow of ecosystem goods and services.

co-chaired by Nobel Prize winners *Kenneth Arrow* and *Robert Solow* to examine the issue. In January 1993, the panel issued a report which concluded that "CV studies can produce estimates reliable enough to be the starting point for judicial or administrative determination of natural resource damages—including lost passive-use value (i.e. non-use value)" (Arrow et al. 1993).

At the same time, the controversy about CV also stimulated a substantial body of transdisciplinary ESV research. Highlights include conjoint analysis, Meta-Analysis (MA), group valuation, and Multiple Criterion Decision Analysis (MCDA), each of which is discussed below.

Insights from psychology have proven fruitful in structuring and interpreting contingent valuation studies (e.g. Kahneman and Knetsch 1992). A new approach, which gained popularity in the 1990s was *conjoint analysis* (e.g. Mackenzie 1992, Adamowicz et al. 1994, Boxall et al. 1996, Hanley 1998). This technique allowed researchers to identify the marginal value of changes in the *characteristics* of environmental resources, as opposed to asking direct CV questions. Respondents are asked to choose the most preferred alternative (or, to rank the alternatives in order of preference, or to rate them on some scale) among a given set of hypothetical alternatives, each depicting a different bundle of environmental attributes. Responses to these questions can then be analyzed to determine the marginal rates of substitution between any pair of attributes that differentiate the alternatives. If one of the characteristics has a monetary price, then it is possible to compute the respondent's willingness to pay for the other attributes.

While subject to the same concern as CV regarding the hypothetical nature of valuation, the conjoint analysis approach offers some advantages (Farber and Griner

2000). For example, it creates the opportunity to determine tradeoffs in environmental conditions through its emphasis on discovering whole preference *structures* and not just monetary valuation. This may be especially important when valuing ecosystems, which provide a multitude of joint goods and services. In addition, it more reasonably reflects multi-attribute choice than the typical one-dimensional CV.

A well-developed approach in psychological, educational, and ecological research, *Meta-Analysis* (MA) was introduced to the ESV field by Walsh and colleagues in the late 1980s and early 1990s (Walsh et al. 1989, Walsh et al. 1992, Smith and Karou 1990). MA is a technique that is increasingly used to understand the influence of methodological and study-specific factors on research outcomes and to synthesize past research. Recent applications include meta-analyses of air quality (Smith and Huang 1995), endangered species (Loomis and White 1996), and wetlands (Brouwer et al. 1997, Woodward and Wui). A more recent use of meta-analysis is the systematic utilization of the existing value estimates from the source literature for the purpose of value transfer (Rosenberger and Loomis 2000, Shrestha and Loomis 2003).

Mainly derived from political theory, *discourse-based valuation* is founded on the principles of deliberative democracy and the assumption that public decision-making should result, not from the aggregation of separately measured individual preferences, but from a process of open public debate (Jacobs 1997, Coote and Lenaghan 1997). This method is extremely useful in ESV as it addresses the fairness goal we mentioned earlier because ecosystem services are very often public goods (e.g. global climate regulation, biodiversity) that are shared by social groups (Wilson and Howarth 2002; Howarth and Wilson 2006).

MCDA techniques originated over three decades ago in the fields of mathematics and operations research and are well-developed and well-documented (Hwang and Yoon, 1981). These provide a structured framework for decision analysis which involves definition of goals and objectives, identification of the set of decision options, selection of criteria for measuring performance relative to objectives, determination of weights for the various criteria, and application of procedures and mathematical algorithms for ranking options.

Compared to Cost-Benefit Analysis, MCDA has at least these three advantages (Munda 1995): 1) by definition MCDA is multi-dimensional and can consider different and incommensurable objectives (such as sustainability, equity and efficiency) at the same time; 2) MCDA is much more flexible in structure as well as aggregation procedures; (In a hypothetical case all indicators do not have to be valued in monetary terms. Instead, the original measurement units could be kept or normalized in different ways, which makes room for subjective components of the analysis); and 3) MCDA has the capacity to take into account qualitative variables. (This is especially useful when uncertainty is an issue. For instance, the effect of global warming on species diversity is uncertain and could be expressed qualitatively.) Of course, MCDA also has it own limitations such as 1) a multi-criteria problem is by definition mathematically ill-structured i.e. it has no objective solution. This is also the primary reason for the flowering of many different theories and models; 2) various aggregation procedures exist for MCDA, which could be confusing because one method has to be chosen and the final results are very sensitive to this step.

The emergence of these new interdisciplinary methods can be attributed in part to two workshops in the 1990s that brought together ESV researchers from different disciplines (EPA 1991 and NCEAS 1999, summarized in special issues of *Ecological Economics* in 1995 and 1998 respectively). The organizers of the first workshop believed that "the challenge of improving ecosystem valuation methods presents an opportunity for partnership—partnership between ecologists, economists, and other social scientists and policy communities. Interdisciplinary dialogue is essential to the task of developing improved methods for valuing ecosystem attributes" (Bingham et al. 1995). In a paper comparing economics and ecological concepts for valuing ecosystem services, participants from the second workshop concluded that "there is clearly not one 'correct' set of concepts or techniques. Rather there is a need for conceptual pluralism and thinking 'outside the box'" (Farber et al. 2002).

This call for cross-disciplinary research is echoed by a recent National Research Council (NRC) study on assessing and valuing the ecosystem services of aquatic and related terrestrial ecosystems. In their final report a team composed of 11 experts from the fields of ecology, economics, and philosophy offered guidelines for ESV including: "Economists and ecologists should work together from the very beginning to ensure the output from any/ an ecological model is in a form that can be used as input for an economic model" (National Research Council 2005). Their prepublication version of the report titled "*Valuing ecosystem services: toward better environmental decision-making"* is available online at http://books.nap.edu/books/030909318X/html

Two interdisciplinary publications drew widespread attention to ecosystem service valuation and stimulated a continuing controversy between ecological economists and traditional "neoclassical" economists. Costanza and his colleagues (ecologists and economists) published an often-cited paper in *Nature* on valuing the services provided by global ecosystems. They estimated that the annual value of 17 ecosystem services for the entire biosphere was US$33 trillion (Costanza et al. 1997). The journal *Ecological Economics* contributed a special issue in 1998, which included a series of 13 commentaries on the *Nature* paper.

The first book dedicated to ecosystem services was also published in 1997 (Daily et al. 1997). *Nature's Services* brought together world-renowned scientists from a variety of disciplines to examine the character and value of ecosystem services, the damage that has been done, and the consequent implications for human society. Contributors including Paul R. Ehrlich, Donald Kennedy, Pamela A. Matson, Robert Costanza, Gary Paul Nabhan, Jane Lubchenco, Sandra Postel, and Norman Myers present a detailed synthesis of the latest understanding of a suite of ecosystem services and a preliminary assessment of their economic value.

Starting in April 2001, more than 2,000 experts have been involved in a four-year effort to survey the health of the world's ecosystems and the threats posed by human activities. The Millennium Assessment has fundamentally changed the landscape in ecosystem service research by switching attention from ecological processes and function to the service itself (Perrings 2006). The synthesis report is now available for review at http://www.millenniumassessment.org/en/index.aspx

ESV in Practice

In the ESV area most of the final demand comes from policy makers and public agencies[8]. To what extent, however, is ESV actually used to make real environmental decisions?

The answer to this question is contingent on the specific areas of environmental policy which are of concern. There are a few areas in which ESV is well established. These include Natural Resource Damage Assessment (NRDA) cases in the USA, CBA of water resource planning, and planning for forest resource use (Adamowicz 2004). In other areas, however, there have been relatively few documented applications of ESV where it was used as the sole or even the principal justification for environmental decisions, and this is especially true in the natural resources planning area (cf. McCollum 2003 for some examples though).

A number of factors have limited the use of ESV as a major justification for environmental decisions. These include methodological problems that affect the credibility of the valuation estimates, legislative standards that preclude consideration of cost-benefit criteria, and lack of consensus about the role that efficiency and other criteria should play in the design of environment regulations (see later section for details on debates on ESV). However, while environmental decisions may not always be made solely or mainly on the basis of net benefits, ESV has a strong influence in stimulating awareness of the costs and gains stemming from environmental decisions, and often plays a major role in influencing the choice among competing regulatory alternatives (Froehlich et al. 1991).

[8] Reviews of the use of ESV in policy include Navrud and Pruckner (1997), Bonnieuz and Rainelli (1999), Loomis (1999), Pearce and Seccombe-Hett (2000), Silva and Pagiola (2003), McCollum (2003) and Adamowicz (2004).

In Europe, the history of both research and applied work in ESV is much shorter than in the U.S.A. Usually, environmental effects are not valued in monetary terms within the European Union. In a number of European countries CBA has been used as a decision tool in public work schemes, especially in road construction (Navrud and Prukner 1997). In earlier years, environmental policy at the European Union level was not informed by environmental appraisal procedures, where appraisal is taken to mean a formal assessment of policy costs and effectiveness using *any* established technique including ESV. But this picture has changed in recent years, and the use of ESV is now accelerating as procedures for assessing costs and benefits are introduced in light of changes to the Treaty of Union (Pearce and Seccombe-Hett, 2000).

A recent report from the World Bank provides a positive view of the use of ESV in the form of CBA in World Bank projects (Silva and Pagiola 2003). The results show that the use of CBA has increased substantially in the last decade. Ten years ago, one project in 162 used CBA. By contrast, as many as one third of the projects in the environmental portfolio did so in recent years[9] While this represents a substantial improvement, the authors predicted "there remains considerable scope for growth" (p1).

Next we will focus on ESV's roles in (1) Natural Resource Damage Assessments (NRDA), (2) CBA/CEA (Cost Effectiveness Analysis), and (3) natural capital accounting. Because there are no specific mechanisms that track the process of how and when research becomes policy, we have to rely on examples and, therefore, offer an anecdotal overview.

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[9] An examination of the types of valuation methods used in these World Bank studies shows that market based methods such as avoided costs and changes in productivity are far more common than are contingent valuation, hedonic price, or other ESV methodologies (Silva and Pagiola 2003).

ESV in NRDA

NRDA is the process of collecting, compiling, and analyzing information to determine the extent of injuries to natural resources from hazardous substance releases or oil discharges, and to determine appropriate ways of restoring the damaged resources and compensating for those injuries (cf. Department of Interior (DOI) Natural Resource Damage Assessments 1980 and Department of Commerce Natural Resource Damage Assessments 1990). Two environmental statutes provide the principle sources of federal authority over natural resource damages: the *Comprehensive Environmental Response, Compensation, and Liability Act* (CERCLA) and the *Oil Pollution Act* (OPA). Although other examples of federal legislation addressing natural resource damages do exist, these two statutes are the most generally applicable and provide a consistent framework in which to discuss natural resource damage litigation.

Under the DOI regulations, valuation methodologies are used to calculate "compensable values" for interim lost public uses. Valuation methodologies include both market-based methods (*e.g.,* market price and/or appraisal) and non-market methodologies (*e.g.,* factor income, travel cost, hedonic pricing, and contingent valuation). Under the OPA, trustees for natural resources base damages for interim lost use on the cost of "compensatory restoration" actions. Trustees can determine the scale of these actions through methodologies that measure the loss of services over time or through valuation methodologies. In any case NRDA poses a big challenge for ESV as a dollar value estimate of total damages is required and valuing multiple ecosystem services typically multiplies the difficulty of evaluation.

Although statutory authorities existed prior to the 1989 *Exxon Valdez* oil spill, the spill was a singular event in the development of trustee NRDA programs. In the years following the spill, NRDA has been on the forefront of ESV use in litigation. The prospect of extensive use of non-market methods in NRDA has generated extensive controversy, particularly among potentially responsible parties (cf. Hanemann, 1994, and Diamond and Hausman, 1994, for differing viewpoints on the reliability of the use of contingent valuation in NRDA as well as in CBA in general).

In the Exxon Valdez case, a team of CV researchers was hired by the State of Alaska to conduct a study of the lost "passive use value" caused by the spill, and the team produced a conservative assessment of 2.8 billion dollars (Carson 1992). Exxon's own consultants published a contrasting critical account of CV arguing that the method cannot be used to estimate passive-use values. Their criticism mainly focused on situations where respondents have little experience using the ecosystem service that is to be altered and when the source of the economic value is not the result of some in site use (Hausman 1993)[10].

This argument led to the previously mentioned NOAA panel, which after a lengthy public hearing and review of numerous written submissions issued a report that cautiously accepted the reliability of CV (Arrow et al. 1993).

In the context of the wide-ranging public debate that continued after the Exxon Valdez case, NOAA reframed the interim lost value component from a monetary compensation measure (*how much money does the public require to make it whole?*) to a resource compensation measure (*how much compensatory restoration does the public*

[10] Much of this debate could be reconciled if the critiques distinguished concerns about the CV itself from a belief that CV estimates do not measure economic values because they are not the result of an economic choice (Smith 2000).

require to make it whole?). By recovering the costs of compensatory restoration actions (costs of resource compensation) rather than the value of the interim losses (monetary compensation), the revised format deflects some of the public controversy about economic methods (Jones and Pease 1997). However, some researchers argue, for instance, that money cannot be removed from NRDA for the simple reason that failure to consider money leaves trustees unable to judge the adequacy of compensating restoration (Flores and Thacher 2004).

ESV in a CBA-CEA framework

CBA is characterized by a fairly strict decision-making structure that includes defining the project, identifying impacts which are economically relevant, physically quantifying impacts as benefits or costs, and then calculating a summary monetary valuation (Hanley and Spash 1993). CEA has a rather similar structure, although only the costs of alternative means of achieving a previously defined set of objectives are analyzed. CBA provides an answer to "whether to do", and CEA answers "how to do".

When the Reagan administration came to power it attempted to change the role of government in the private affairs of households and firms. Regulatory reform was a prominent component of its platform. President Reagan's Executive Order No. 12291 requiring a CBA for all new major regulations whose annual impact on the economy was estimated to exceed $100 million (Smith 1984). The aim of this Executive Order was to develop more effective and less costly regulation. It is believed that the impact of EO 12291 fell disproportionately on environmental regulation (Navrud and Pruckner 1997).

President Bush Sr used the same Executive Order. President Clinton issued Executive Order 12866, which is similar to Reagan's order but changes some requirements. The order requires agencies to promulgate regulations if the benefits "justify" the costs. This language is generally perceived as more flexible than Reagan's order, which required the benefits to "outweigh' the costs. Clinton's order also places greater emphasis on distributional concerns (Hahn 2000).

CBA analysis for environmental rule making under the George W. Bush administration remains controversial. At the core of the controversy is the growing influence of the White House office with responsibility for cost-benefit review: the Office of Information and Regulatory Affairs (OIRA), within the Office of Management and Budget (OMB). Traditionally, OIRA has had fairly minimal interaction with submitting agencies as they prepare cost-benefit analyses. But under its current administrator, John Graham, OIRA has become intimately involved in all aspects of the cost-benefit process. During the eight years of the Clinton administration, OIRA sent 16 rules back to agencies for rewriting. Graham sent back 19 rules (not all of which were environmental) during his first year alone.

Originally, CBAs reflected mainly market benefits such as job creation and added retail sales. More recently, attempts have been made to incorporate the environmental impacts of projects/policies within CBA to improve the quality of government decision-making. The use of ESV allows CBA to be more comprehensive in scope by incorporating environmental values and putting them on the same footing as traditional economic values.

EPA's National Center for Environmental Economics' online library is a good resource for all CBAs conducted over the years. The most common ESV application by the EPA involves analyses of the benefits of specific regulations as part of Regulatory Impact Analyses (RIAs). Although RIAs—and hence ESV—have been performed for numerous rules, the scope and quality of the ESV in these RIAs has varied widely. A review of 15 RIAs performed by the EPA between 1981 and 1986 (EPA and OPA 1987) found that only six of the 15 RIAs addressed by the study presented a complete analysis of monetized benefits and net benefits. The 1987 study notes that many regulations were improved by the analysis of benefits and costs, even where benefits were not monetized and net benefits were not calculated.

One famous example of the use of CEA is the 1996 New York Catskills Mountains Watershed case where New York City administrators decided that investment in restoring the ecological integrity of the watershed would be less costly in the long-run than constructing a new water filtration plant. New York City invested between $1 billion and $1.5 billion in restoratory activities in the expectation of realizing cost savings of $6 billion–$8 billion over 10 years, giving an internal rate of return of 90–170% and a payback period of 4–7 years. This return is an order of magnitude higher than is usually available, particularly on relatively risk-free investments (Chichilnsky and Heal 1998).

ESV in natural capital accounting

Though closely related, "Green" GDP accounting and natural capital accounting are different. GDP aggregates all sources of well-being, including all market goods and services, into a single index. Green GDP adds missing ecological elements to conventional GDP by including non-market contributions to welfare. Natural capital accounting usually separately accounts for *all* nature's contributions to welfare, including those captured in GDP as intermediate products such as pollination's contribution to increased agricultural output. Proposals have been made to integrate the results of natural capital accounting into Green GDP though researchers have cautioned against double accounting and the simple add-up approach (Boyd and Banzhaf 2006). So far there have been a handful of studies that attempted to plug ecosystem service valuation results into Green GDP accounting (Gren 2003; Matero and Saastamoinen 2007), for example, by using the supply side of the Input-Output model (Gret-Regamey and Kytzia 2007) to avoid double accounting.

For the purpose of this paper we'll only focus on natural capital accounting, which was popularized by the effort to value the ecosystem services and natural capital at the global scale (Costanza et al. 1997). Since then there have been numerous studies to value natural capital at a national level (e.g. Anielski and Wilson 2005) and at the state/regional level (e.g. Wilson and Troy 2003, Anielski and Wilson 2005, Asafu-Adjaye et al. 2005, Costanza et al. 2007). Attempting to include the value of all ecosystem services, these studies used benefit transfer of results from the empirical valuation literature. A couple of resent trends are to combine the transferred results with Geographical Information Systems (GIS) (cf. Troy and Wilson 2006 for a review) and ecosystem modeling.

GIS has been used to increase the context specificity of value transfer (e.g. Eade and Moran 1996, Wilson et al. 2004). In doing so, the value transfer process is augmented with a set of spatially explicit factors so that geographical similarities between the policy site and the study site are more easily detected. In addition, the ability to present and calibrate economic valuation data in map form offers a powerful means for expressing environmental and economic information on multiple scales to stakeholders.

Thanks to the increased ease of using Geographic Information Systems (GIS) and the public availability of land cover data sets derived from satellite images, ecosystem service values can more easily be attributed to geographical locations and areas. In simplified terms, the technique involves combining one land cover layer with another layer representing the geography by which ecosystem services are aggregated - i.e. watershed, town or park. ESV is made spatially explicit by disaggregating landscapes into their constituent land cover elements and ecosystem service types (Wilson et al. 2004). Spatial disaggregation increases the potential management applications for ecosystem service valuation by allowing users to visualize the explicit location of ecologically important landscape elements and overlay them with other relevant themes for analysis. Disaggregation is also important for descriptive purposes, for the pattern of variation is often much more telling than any aggregate statistic.

In order for stakeholders to evaluate the change in ecosystem services, they must be able to query ecosystem service values for a specific and well-defined area of land that is related to an issue pertinent to them. For this reason, several types of spatially-explicit boundary data can be linked to land cover and valuation data within a GIS. The aggregation units used for ecosystem service mapping efforts should be driven by the intended policy or management application, keeping in mind that there are tradeoffs to reducing the resolution too much. For example, a local program targeted at altering land management for individual large property owners might want to use individual land parcel boundaries as the aggregation unit. However, such a mapping level would yield far too much information for national-level application. A state agency whose programs affect all lands in the state (e.g. a water resources agency) might use watersheds as units or a state agency managing state parks might be better off using the park boundaries, or park district boundaries as units.

For example, The EcoValue Project draws from recent developments in ecosystem service valuation, database design, internet technology, and spatial analysis techniques to create a web-accessible, GIS decision support system. The site uses empirical studies from the published literature that are then used to estimate the economic value of ecosystem services (cf. http://ecovalue.uvm.edu). Using watersheds as the primary unit of spatial aggregation, the project provides ecosystem service value estimates for the State of Maryland and the four state Northern Forest region including New York, Vermont, New Hampshire and Maine. The end result is a GIS value-transfer platform that provides the best available valuation data to researchers, decision-makers, and public stakeholders throughout the world.

In a study of the Massachusetts landscape using a similar technique, Wilson and colleagues (Wilson et al. 2004) found that the annual non-market ecosystem service value was over $6.3 billion annually for the state. As in many areas, most development in Massachusetts has come at the expense of forest and agricultural land. Based on the net forest and agricultural land lost to all forms of development between 1985 and 1999, an *ex post* study showed that the state lost over $200 million *annually* in ecosystem service value during the period, based on 2001 US dollars. Had the same amount of development occurred in a way that impacted less forest and agricultural land through denser "in-fill" development and more brownfield development, the state could have enjoyed the economic benefits of both development and ecosystem services (Wilson and Troy 2003).

Recognizing the value of ecosystem services, decision-makers have started to adopt *ex ante* ESV research linked with computer modeling. An example of this was an integrated modeling and valuation study of fynbos ecosystems in South Africa (Higgins et al. 1997). In this example, a cross-section of stakeholders concerned about the invasion of fynbos ecosystems by European pine trees worked together to produce a simulation model of the dynamics and value of the ecosystem services provided by the system. The model allowed the user to vary assumptions and values for each of the services and observe the resulting behavior and value of the ecosystem services from the system. This model was subsequently used by park managers to design (and justify) containment and removal efforts for the pine trees.

In a more recent example, the city of Portland's Watershed Management Program sponsored a Comparative Valuation of Ecosystem Services (CVES) analysis in order to understand the tradeoffs between different flood control plans. Integrated with ecosystem modeling, an ESV study under CVES showed that a proposed flood abatement project in the Lent area could provide more than $30,000,000 in benefits (net presented value) to the public over a 100-year timeframe. Five ecosystem services would increase productivity as a result of floodplain function improvements and riparian restoration (David Evans and Associates Inc. and EcoNorthwest 2004).

Modeling has also been combined with GIS to understand and value the spatial dynamics of ecosystem services. An example of this application was a study of the 2,352 km^2 Patuxent river watershed in Maryland (Bockstael et al. 1995, Costanza et al. 2002).

This model was used to addresses the effects of both the magnitude and spatial patterns of human settlements and agricultural practices on hydrology, plant productivity, and nutrient cycling in the landscape, and the value of ecosystem services related to these ecosystem functions. Several historical and future scenarios of development patterns were evaluated in terms of their effects on both the biophysical dynamics of ecosystem services and the value of those services. A recent effort is to use spatially-explicit dynamic modeling to integrate our understanding of ecosystem functioning, ecosystem services, and human well-being across a range of spatial scales (http://www.uvm.edu/giee/?Page=research/ecosystemservices/index.html).

Debate on the use of ESV

There are multiple policy purposes and uses of ESV. These uses include:

1. to provide for comparisons of natural capital to physical and human capital in regard to their contributions to human welfare.
2. to monitor the quantity and quality of natural capital over time with respect to its contribution to human welfare
3. to provide for evaluation of projects that propose to change (enhance or degrade) natural capital.

Much of the debate about the use of ESV has to do with not appreciating this range of purposes. In addition there are a range of other obstacles and objections to the use of ESV. In summarizing experiences of ESV use from six countries, Barde and Pearce (1991) mentioned three main categories of obstacles: (1) ethical and philosophical, (2) political, and (3) methodological and technical. Below we discuss each of these in greater detail.

Ethical and philosophical debate

Ethical and philosophical obstacles arise from a criticism of the conventional welfare economics foundations of ESV. In particular, “monetary reductionism”, illustrated by the willingness-to-pay criterion, is strongly rejected in “deep ecology” circles or by those who claim that ecosystems are not economic assets and that it is therefore immoral to measure them in monetary terms (e.g. Norgaard et al. 1998, McCauley 2006). Based exclusively on an individual’s preferences, the principle of utility maximization is judged to be too reductionist a basis on which to make decisions involving environmental assets, irreversibility and future generations (Vatn and Bromley 1994, Matinez-Alier et al. 1998).

Practitioners of ESV argue that the ESV concept is much more complex and nuanced than these objections acknowledge. Monetization is simply a convenient means of expressing the relative values that society places on different ecosystem services. If these values are presented solely in physical terms—so much less provision for clean water, perhaps, and so much more production of crops—then the classic problem of comparing apples and oranges applies. The purpose of monetary valuation is to make the disparate services provided by ecosystems comparable to each other, using a common metric. Alternative common metrics exist (including energy units and land units i.e. the “ecological footprint”) but in the end, the choice of metric is not critical because, given appropriate conversion factors, one could always translate results of the underlying trade-offs from one metric to another.

The key issue here comes down to trade-offs. *If* one does not have to make tradeoffs between ecosystem services and other things, *then* valuation is not an issue. *If* however, one does have to make such tradeoffs, *then* valuation will occur, whether it is explicitly recognized or not (Costanza et al. 1997). Given this, it seems better that the trade-offs be made explicit.

The usefulness lies in the fact that ESV uses easily understood and accepted rules to reduce complex clusters of effects and phenomena to single-valued commensurate magnitudes, that is, to dollars. The value of the benefit-cost framework lies in its ability to organize and simplify certain types of information into commensurate measures (Arrow et al. 1996).

While we believe that there is a strong case in favor of monetary valuation as a decision aid to help make trade-offs more explicit, we also recognize that there are limits to its use. Expanding ESV towards sustainability and fairness goals (on top of the traditional efficiency goal) will help expand the boundaries of those limits (Costanza and Folke 1997). A MCDA system that incorporates the triple goals might appear to alleviate the limitations of monetary valuation, but in fact it does not. If there are real trade-offs in the system, those trade-offs will have to be evaluated one way or the other. A MCDA facilitates greater public participation and collaborative decision-making, and allows consideration of multiple attributes (Prato 1999) but it does not eliminate the need to assess trade-offs, and, as we have said, conversion to monetary units is only one way of expressing these trade-offs and all forms of value may and should ultimately contribute to decisions regarding the environment (Costanza 2006).

Political debate

The very objective and virtue of ESV is to make policy objectives and decision criteria explicit, e.g. what are the actual benefits of a given course of action? What is the best alternative? Is the government making an efficient use of environmental resources and public funds? Introducing a public debate on such issues is often unattractive to technical experts and decision-makers and may significantly reduce their margin of action and decision autonomy. Therefore, there may be some reluctance to introduce ESV into political or regulatory debates[11].

Notwithstanding this, humans have to make choices and trade-offs concerning ecosystem services, and, as mentioned above, this implies and requires "valuation" because any choice between competing alternatives implies that the one chosen was more highly "valued." Practitioners of ESV argue that society can make better choices about ecosystems if the valuation issue is made as explicit as possible. This means taking advantage of the best information we can muster, making the uncertainties in that information explicit, and developing new and better ways to make good decisions in the face of these uncertainties. Ultimately, it means being explicit about our goals as a society, both in the short and the long term, and understanding the complex relationships between current activities and policies and their ability to achieve these goals (Costanza 2000).

As Arrow and colleagues (1996) argued, valuation should be considered as a framework and a set of procedures to help organize available information. Viewed in this

[11] This requires ESV researchers to do more than simply develop good ideas to influence policy. They need to understand how the political process affects outcomes, and actively market the use of appropriate and feasible methodologies for promoting environmental policy. In other words, ESV research has to become more problem-driven rather than tool-driven (Hahn 2000).

light, benefit-cost analysis does not dictate choices, nor does it replace the ultimate authority and responsibility of decision makers. It is simply a tool for organizing and expressing certain kinds of information from a range of alternative courses of action. The usefulness of value estimates must be assessed in the context of this framework for arraying information (Freeman 2003).

The more open decision makers are about the problems of making choices and the values involved, and the more information they have about the implications of their choices, the better their choices are likely to be.

Methodological and technical debate

ESV has also been criticized on methodological and technical grounds. There are a range of issues here which are covered in detail elsewhere (e.g. Costanza et al. 1998, Bockstael et al. 2000). For the purposes of this discussion, we will focus on two major issues that seem to underlie much of the debate: purpose and accuracy.

One line of criticism has been that ESV can only be used to evaluate *changes* in ecosystem service values. For example, Bockstael et al. (2000) contended that assessing the total value of global, national, or state level ecosystem services is meaningless because it does not relate to *changes* in services and one would not really consider the possibility of eliminating the entire ecosystem at these scales. But, as mentioned earlier, there are at least three purposes for ESV, and this critique has to do with confusing purpose #3 (assessing changes) with purpose #1 (comparing the contributions of natural capital to human welfare with those of physical and human capital).

To better understand this distinction, the following diagram figure is helpful:

[Insert Figure 5]

The Demand for Services reflects the Marginal Valuations of increasing service levels. The Quantity of Services available determines the Average Valuation of that service over its entire range. Consequently, Average Value x Quantity would represent a "Quasi-Market Valuation" of that service level. In a restricted sense, if there were a market for the service, this would be the revenue obtained from the service, comparable to an indicator like the sales volume of the retail sector. It would be directly comparable and analogous to the valuation of income flows from physical capital, and could be capitalized to reflect the market value of natural capital and compared to similarly capitalized values for physical investment. Furthermore, changes in the volume or value of this service could be capitalized to reflect the value of new natural capital investment/disinvestment, just as we measure new investment and depreciation in physical capital at the macro level (Howarth and Farber 2002)

This "Quasi-market value" has a restricted meaning. Of course, it does not reflect the "full value" of the service to human welfare because full value is the sum of marginal values; i.e., the area under the demand curve. However, the more substitutes there are available for the service, the less the difference between "full value" and this quasi-market value. In addition, this quasi-market value is more directly comparable with the quasi-market value of the physical and human capital contributors to human welfare as measured in aggregate indicators like GDP. So, if ones purpose is to compare contributions of natural capital to human welfare with those of physical and human capital (as estimated in GDP, for example) then this is an appropriate (albeit not perfect) measure.

Furthermore, if there really were a market for the service, and economies actually had to pay for it, the entire economics of many markets directly or indirectly impacted by the service would be altered (Costanza et al. 1998). For example, electricity would become more costly, altering its use and the use of energy sources, in turn altering the costs and prices of energy using goods and services. The changes in markets would likely feedback on the demand for the ecosystem service, increasing or decreasing it, depending on the service and its economic implications. The "true market value" could only be determined through full scale ecologic-economic modeling. While modeling of this type is underway (cf. Boumans et al. 2002), it is costly and difficult to do, and meanwhile decisions must be made. "Quasi-market value" is thus a reasonable first order approximation for policy and public discourse purposes if we want to compare the contributions of natural capital to the contributions of other forms of capital to human welfare.

ESV can also be used to assess the impact of specific changes or projects. Balmford et al. (2002) is a recent example of this use of ESV at the global scale. In this study, the costs and benefits of expanding the global nature reserve network to encompass 15% of the terrestrial biosphere and 30% of the marine biosphere were evaluated, concluding that the benefit-cost ratio of this investment was approximately 100:1. In these circumstances, Average Value x ΔQ is likely to be a reasonable measure of the economic value of the change in services; an overestimate of benefits for service increases, and an underestimate of costs for service decreases. The degree of over- or under-estimation depends again on the replaceability of the service being gained or lost.

Beyond the confusion concerning purposes, the *accuracy* of ESV is also sometimes questioned. Diamond and Hausman (1994), for instance, asked the question, "[In] *contingent valuation--is some number better than no numbers?*"

In our view, the answer to this question also depends on the intended use of the ESV result and the corresponding accuracy required (Brookshire and Neill 1992, Desvousges et al. 1992). As Figure 6 shows we can think of accuracy as existing along a continuum whereby the minimum degree of accuracy needed is related to the cost of making a wrong decision based on the ESV result.

[Insert Figure 6]

For example, using ESV to assist an environmental policy decision-maker in setting broad priorities for assessment and possible action may require a moderate level of accuracy. In this regard, any detriment resulting from minor inaccuracies is adequately offset by the potential gains. This use of ESV represents an increase of knowledge that costs society relatively little if the ESV results are later found to be inaccurate. However, if ESV is used as a basis for a management decision that involves irreversibility, the costs to society of a wrong decision can be quite high. In this case, it can be argued that the accuracy of a value transfer should be very high.

Findings and directions for the future

ESV is often complex, multi-faceted, socially contentious and fraught with uncertainty. In contrast, traditional ESV research involves the work of experts from separate disciplines, and these studies often turn out to be overly simple, uni-dimensional and "value-free". Our survey of the literature has shown that over time, there has been movement toward a more transdisciplinary approach to ESV research that is more consistent with the nature of the problems being addressed.

The truly *transdisciplinary* approach ultimately required for ESV is one in which practitioners must accept that disciplinary boundaries are academic constructs that are irrelevant outside of the university, and must also allow the problem being studied to determine the appropriate set of tools, rather than vice versa.

What is needed are ESV studies that encompass all the components mentioned in Figure 1 earlier, including ecological structures and processes, ecological functions, ecosystem services, human welfare, land use decisions, and the dynamic feedbacks between them. To our knowledge, there have been few such studies to date. But it is just this type of study that is of greatest relevance to decision-makers and it looks to be the way forward (Turner et al. 2003).

Figure 7 indicated how little effort has gone into understanding the linkages between ecological functions, services, and human welfare. Among 675 peer-reviewed ESV studies (with a total of 730 data points) published in the past 35 years, most effort has gone into the understanding of human preferences for ecosystem services that are directly consumed, including 34% valuing recreation benefits and 18% valuing water quality change. In comparison, most supporting and regulating services are undervalued if they are valued at all.

[Insert Figure 7]

Obviously there has been great progress in ecology and in understanding ecosystem processes and functions, and in the economics of developing and applying non-market techniques for valuation, however there remains a gap between the two. To quote a recent ESV report by an inter-disciplinary group of ecologists, economists, and philosophers, ‘‘…the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values’’ (National Research Council 2005, p. 2).

Nevertheless, some useful integrated studies are starting to emerge to bridge the gap between ecosystem functions and services , including those valuing biological control (Cleveland et al. 2006) and pollination services (Ricketts et al. 2004, Olschewski et al. 2006, Priess et al. 2007).

This paper also attempted to quantify ESV's contribution to environmental policy-making by answering questions such as "to what extent is ESV actually used to make real decisions?" However, it was soon realized that this goal was too ambitious. Instead, along with other reviewers (e.g. Pearce and Seccombe-Hett 2000, Adamowicz 2004), it was found that the contribution of ESV to ecosystem management has not been as large as hoped or as clear as imagined, although it is widely used in NRDA, CBA-CEA, and natural capital accounting.

We discussed the three types of obstacles to the use of ESV in policy making. While there is a strong case in favor of monetary valuation as a decision-aid, we also recognize that there are limits to its use. These limitations are due to the complexity of both ecological systems and values, which could be more adequately incorporated by the triple-goal ESV system. Valuing ecosystem services with not only efficiency, but also fairness and sustainability as goals, is the next step needed to promote the use of ESV in ecosystem management and environmental policy making. This new system can be well supported by current transdisciplinary methodologies such as participatory assessment (Campell and Luckert 2002), group valuation (Jacobs 1997, Wilson and Howarth 2002, Howarth and Wilson 2006), and the practice of integrating ESV with GIS and ecosystem modeling (Bockstael et al. 1995, Costanza et al. 2002, Boumans et al. 2002).

ACKNOWLEDGEMENTS

This work was supported in part by Contract No. SR04-075, "Valuation of New Jersey's Natural Capital" from the New Jersey Department of Environmental Protection, William Mates, Project Officer.

Box 1: Ecosystem service valuation methods (adapted from Farber et al. 2006)

Conventional economic valuation

Revealed reference approaches

- Market methods: Valuations are directly obtained from what people must be willing to pay for the service or good (e.g., timber harvest).
- Travel cost: Valuations of site-based amenities are implied by the costs people incur to enjoy them (e.g., cleaner recreational lakes).
- Hedonic methods: The value of a service is implied by what people will be willing to pay for the service through purchases in related markets, such as housing markets (e.g., open-space amenities).
- Production approaches: Service values are assigned from the impacts of those services on economic outputs (e.g., increased shrimp yields from increased area of wetlands).

State-reference approaches

- Contingent valuation: People are directly asked their willingness to pay or accept compensation for some change in ecological service (e.g., willingness to pay for cleaner air).
- Conjoint analysis: People are asked to choose or rank different service scenarios or ecological conditions that differ in the mix of those conditions (e.g., choosing between wetlands scenarios with differing levels of flood protection and fishery yields).

Cost-based approaches

- Replacement cost: The loss of a natural system service is evaluated in terms of what it would cost to replace that service (e.g., tertiary treatment values of wetlands if the cost of replacement is less than the value society places on tertiary treatment).
- Avoidance cost: A service is valued on the basis of costs avoided, or of the extent to which it allows the avoidance of costly averting behaviors, including mitigation (e.g., clean water reduces costly incidents of diarrhea).

Benefit transfer: The adaptation of existing ESV information or data to new policy contexts that have little or no data (e.g. ecosystem service values obtained by tourists viewing wildlife in one park used to estimate that from viewing wildlife in a different park).

Nonmonetizing valuation or assessment

Individual index-based method, including rating or ranking choice models, expert opinion.

Group-based methods, including voting mechanisms, focus groups, citizen juries, and stakeholder analysis.

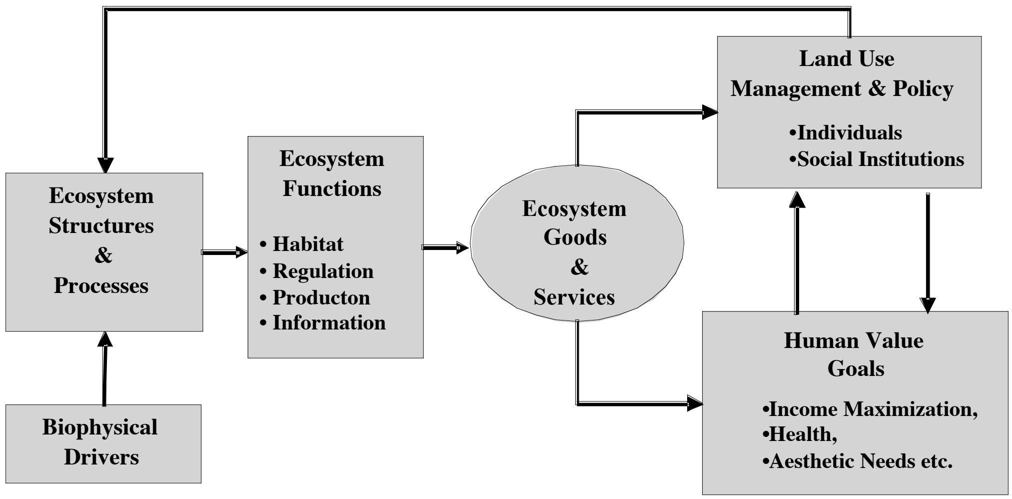

Figure 1: Framework for integrated assessment and valuation of ecosystem goods and services (from de Groot et al. 2002)

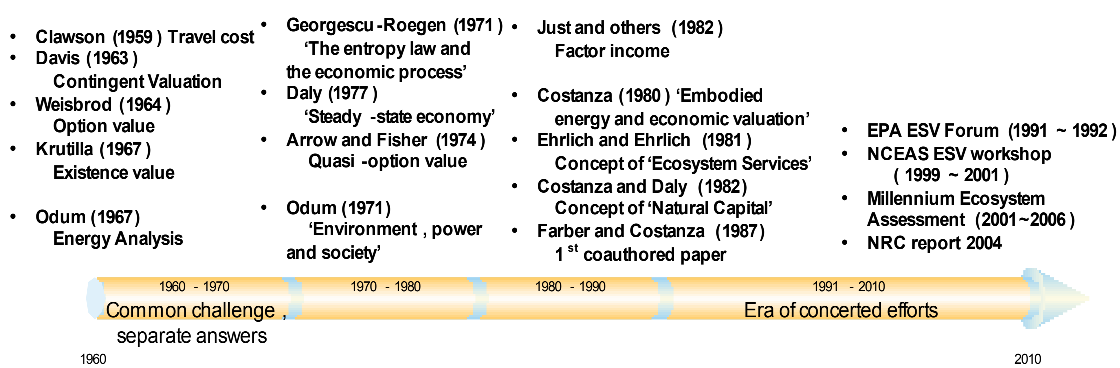

Figure 2: Milestones in the history of ecosystem service valuation

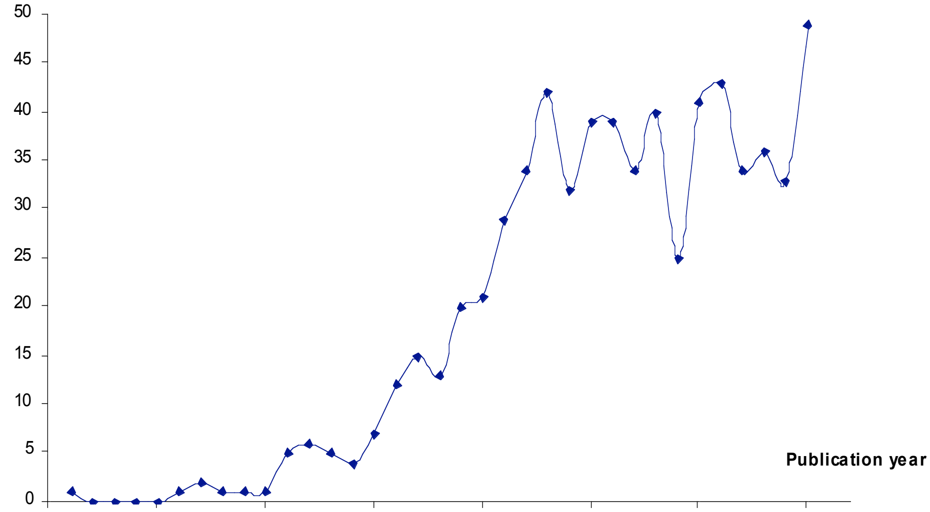

Figure 3: Number of ESV publications in EVRI over time (accessed Feb 10, 2007)

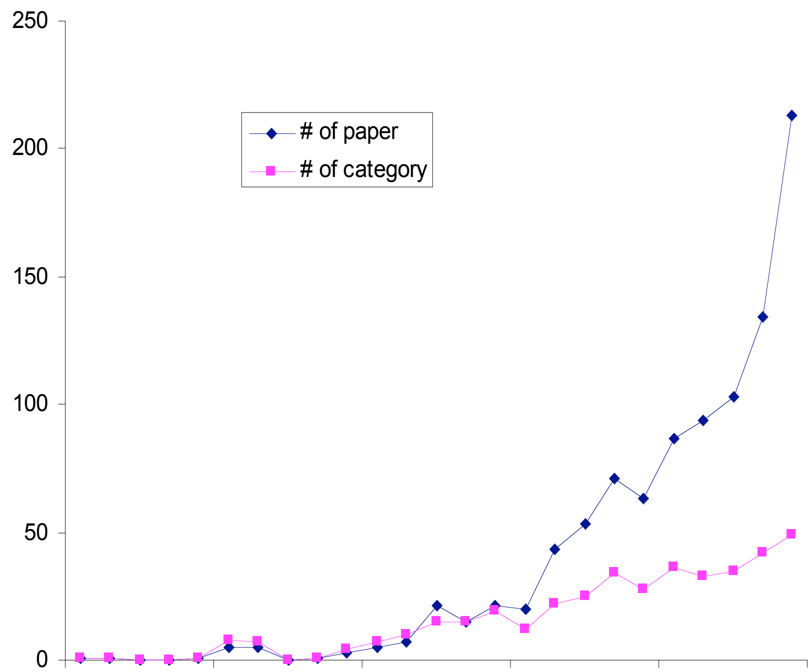

Figure 4: Number of peer-reviewed ecosystem service papers and their related sub-categories over time listed in the ISI Web of Science (accessed June 29, 2007)

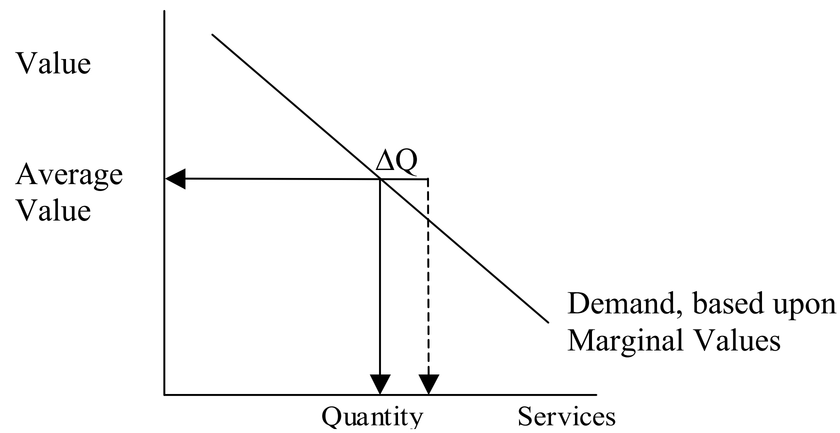

Figure 5: A model of ecosystem service valuation

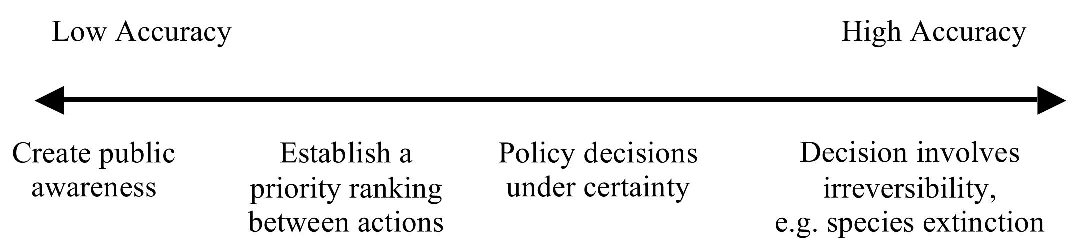

Figure 6: Accuracy Continuum for the ESV (adapted from Desvousges and Johnson 1998)

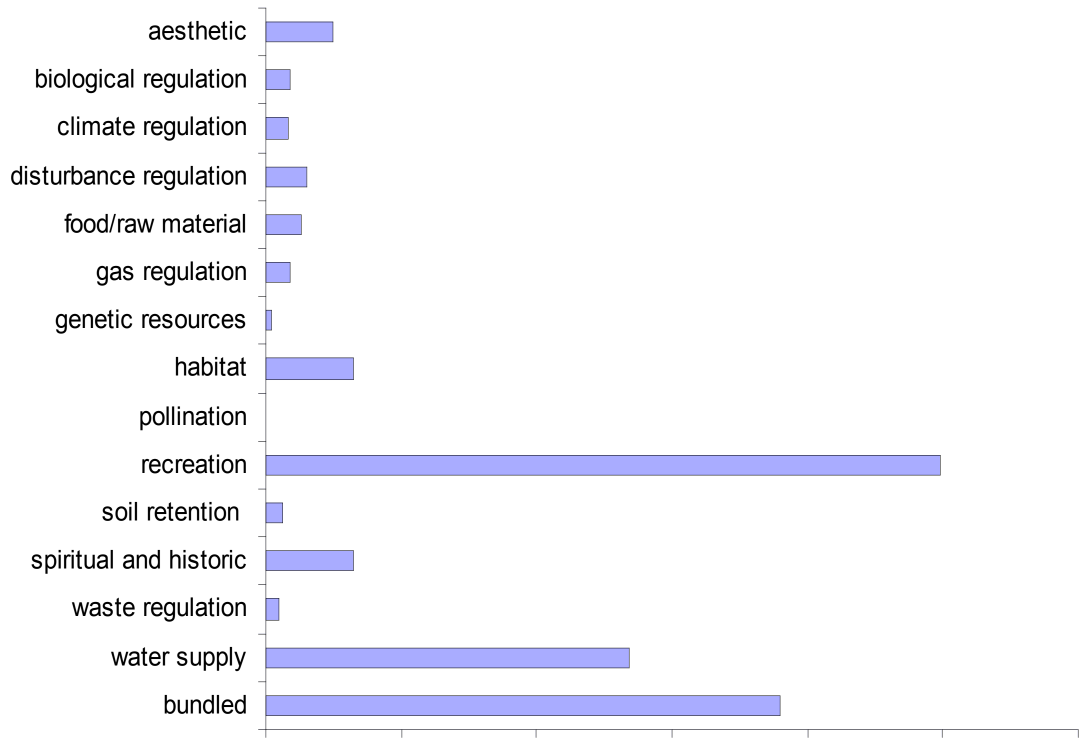

Figure 7: EVRI peer-reviewed valuation data by ecosystem services (total data point = 730, accessed Feb 10, 2007)

Table 1: Categories of ecosystem services and economic methods for valuation (from Farber et al. 2006)

Ecosystem service	Amenability to economic valuation	Most appropriate method for valuation	Transferability across sites
Gas regulation	Medium	CV, AC, RC	High
Climate regulation	Low	CV	High
Disturbance regulation	High	AC	Medium
Biological regulation	Medium	AC, P	High
Water regulation	High	M, AC, RC, H, P, CV	Medium
Soil retention	Medium	AC, RC, H	Medium
Waste regulation	High	RC, AC, CV	Medium to high
Nutrient regulation	Medium	AC, CV	Medium
Water supply	High	AC, RC, M, TC	Medium
Food	High	M, P	High
Raw materials	High	M, P	High
Genetic resources	Low	M, AC	Low
Medicinal resources	High	AC, RC, P	High
Ornamental resources	High	AC, RC, H	Medium
Recreation	High	TC, CV, ranking	Low
Aesthetics	High	H, CV, TC, ranking	Low
Science and education	Low	Ranking	High
Spiritual and historic	Low	CV, ranking	Low

REFERENCES

Adamowicz, W., J. Louviere, et al. (1994). "Combining Revealed and Stated Preference Methods for Valuing Environmental Amenities." Journal of Environmental Economics and Management **26**(3): 271-292.

Adamowicz, W. L. (2004). "What's it worth? An examination of historical trends and future directions in environmental valuation." Australian Journal of Agricultural and Resource Economics **48**(3): 419-443.

Anderson, J. E. (1976). "The social cost of input distortions: a comment and a generalization." American Economic Review **66**(1): 235-238.

Anielski, A. and S. Wilson (2005). Counting Canada's Natural Capital: Assessing the Real Value of Canada's Boreal Ecosystems, The Pembina Institute.

Anielski, M. and S. Wilson (2005). The real wealth of the Mackenzie Region: assessing the natural capital values of a northern boreal ecosystem, Canadian Boreal Initiative.

Arrow, K. and H. Raynaud (1986). Social choice and multicriterion decision-making. Cambridge, MA, MIT Press.

Arrow, K., R. Solow, et al. (1993). Federal Register **58**: 4602-4613.

Arrow, K. J., M. L. Cropper, et al. (1996). "Is There a Role for Benefit-Cost Analysis in Environmental, Health, and Safety Regulation?" Science **272**(5259): 221-222.

Arrow, K. J. and A. C. Fisher (1974). "Environmental Preservation, Uncertainty, and Irreversibility." Quarterly Journal of Economics **88**: 312-319.

Asafu-Adjaye, J., R. Brown, et al. (2005). "On measuring wealth: a case study on the state of Queensland." Journal Of Environmental Management **75**(2): 145-155.

Balmford, A., A. Bruner, et al. (2002). "Ecology - Economic reasons for conserving wild nature." Science **297**(5583): 950-953.

Barbier, E. B. (2007). "Valuing ecosystem services as productive inputs." Economic Policy(49): 178-229.

Barde, J.-P. and D. W. Pearce (1991). Introduction. Valuing the environment: six case studies. J.-P. Barde and D. W. Pearce. London, Earthscan Publications**:** 1-8.

Bingham, G., R. Bishop, et al. (1995). "Issues in Ecosystem Valuation - Improving Information for Decision-Making." Ecological Economics **14**(2): 73-90.

Bishop, R. (1993). "Economic efficiency, sustainability, and biodiversity." Ambio **22**: 69-73.

Bockstael, N., R. Costanza, et al. (1995). "Ecological Economic Modeling and Valuation of Ecosystems." Ecological Economics **14**(2): 143-159.

Bockstael, N. E., A. M. Freeman, et al. (2000). "On measuring economic values for nature." Environmental Science & Technology **34**(8): 1384-1389.

Bonnieux, F. and P. Rainelli (1999). Contingent valuation methodology and the EU institutional framework. Valuing environmental preferences. I. Bateman and M. Williams. New York, Oxford University Press**:** 585-613.

Boulding, K. (1966). The Economics of the Coming Spaceship Earth. Environmental Quality in a Growing Economy. H. Jarrett. Baltimore, John Hopkins Press.

Boumans, R., R. Costanza, et al. (2002). "Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model." Ecological Economics **41**(3): 529-560.

Boxall, P. C., W. L. Adamowicz, et al. (1996). "A comparison of stated preference methods for environmental valuation." Ecological Economics **18**(3): 243-253.

Boyd, J. and S. Banzhaf (2006). What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units. RFF DP 06-02. Washington, D.C., Resources for the future.

Brookshire, D. S. and H. R. Neill (1992). "Benefit Transfers - Conceptual and Empirical Issues." Water Resources Research **28**(3): 651-655.

Brouwer, R., I. H. Langford, et al. (1997). A meta-analysis of wetland contingent valuation studies. Norwich, CSERGE, University of East Anglia.

Campell, B. and M. Luckert, Eds. (2002). Uncovering the Hidden Harvest: Valuation Methods or Woodland and Forest Resources. , . London, Earthscan.

Carson, R. T. (2000). "Contingent valuation: A user's guide." Environmental Science & Technology **34**(8): 1413-1418.

Carson, R. T., M. Hanemann, et al. (1992). A contingent valuation study of lost passive use values resulting from the Exxon Valdez oil spill. Anchorage, Attorney General of the State of Alaska.

Carson, R. T., R. C. Mitchell, et al. (2003). "Contingent valuation and lost passive use: Damages from the Exxon Valdez oil spill." Environmental & Resource Economics **25**(3): 257-286.

Champ, P. A., K. J. Boyle, et al., Eds. (2003). A primer on nonmarket valuation. Dordrecht, The Netherlands, Kluwer Academic Publishers.

Chichilnisky, G. and G. Heal (1998). "Economic returns from the biosphere - Commentary." Nature **391**(6668): 629-630.

Ciriacy-Wantrup, S. (1947). "Capital returns from soil conservation practices." Journal of Farm Economics **29**: 1181-1196.

Clawson, M. (1959). Methods of measuring the demand for and value of outdoor recreation. Washington, D.C., Resource for the Future.

Cleveland, C. J., M. Betke, et al. (2006). "Economic value of the pest control service provided by Brazilian free-tailed bats in south-central Texas." Frontiers in Ecology and the Environment **5**(5): 238-243.

Coote, A. and J. Lenaghan (1997). Citizen juries: theory into practice. London, Institute for Public Policy Research.

Costanza, R. (1980). "Embodied energy and economic valuation." Science **225**: 890-897.

Costanza, R. (2000). "Social goals and the valuation of ecosystem services." Ecosystems **3**(1): 4-10.

Costanza, R. (2006). "Nature: ecosystems without commodifying them." Nature **443**(7113): 749-749.

Costanza, R., R. d'Arge, et al. (1997). "The value of the world's ecosystem services and natural capital." Nature **387**(6630): 253-260.

Costanza, R., R. dArge, et al. (1998). "Auditing the earth: Costanza and his coauthors reply." Environment **40**: 26-27.

Costanza, R. and C. Folke (1997). Valuing ecosystem services with efficiency, fairness and sustainability as goals. Nature's services: societal dependence on natural ecosystems. G. Daily. Washington, D.C., Island Press**:** 49-68.

Costanza, R. and R. A. Herendeen (1984). "Embodied Energy and Economic Value in the United-States-Economy - 1963, 1967 and 1972." Resources and Energy **6**(2):

129-163.

Costanza, R., A. Voinov, et al. (2002). "Integrated ecological economic modeling of the Patuxent River watershed, Maryland." Ecological Monographs **72**(2): 203-231.

Costanza, R., M. Wilson, et al. (2007). The Value of New Jersey's Ecosystem Services and Natural Capital, New Jersey Department of Environmental Protection.

Crocker, T. D. (1999). A short history of environmental and resource economics. Handbook of environmental and resource economics. J. van den Bergh. Cheltenham, U.K., Edward Elgar**:** 32-45.

Daily, G. (1997). Nature's services: societal dependence on natural ecosystems. Washington, D.C., Island Press.

Daly, H. E. (1977). Steady-State Economics. Washington, D.C., Island Press.

Daly, H. E. (1992). "Allocation, distribution, and scale: towards an economics that is efficient, just, and sustainable." Ecological Economics **6**: 185-193.

David Evans and Associates, I. and ECONorthwest (2004). Final submittal: comparative valuation of ecosystem services: Lents project case study.

Davis, R. K. (1963). "Recreation planning as an economic problem." Natural Resources Journal **3**: 239-249.

de Groot, R. S., M. A. Wilson, et al. (2002). "A typology for the classification, description and valuation of ecosystem functions, goods and services." Ecological Economics **41**(3): 393-408.

Desvousges, W. H., M. C. Naughton, et al. (1992). "Benefit Transfer - Conceptual Problems in Estimating Water-Quality Benefits Using Existing Studies." Water Resources Research **28**(3): 675-683.

Diamond, P. A. and J. A. Hausman (1994). "Contingent Valuation - Is Some Number Better Than No Number." Journal of Economic Perspectives **8**(4): 45-64.

Eade, J. D. O. and D. Moran (1996). "Spatial economic valuation: Benefits transfer using geographical information systems." Journal of Environmental Management **48**(2): 97-110.

Ehrlich, P. and A. Ehrlich (1981). Extinction: the causes and consequences of the disappearces of species. New York, Random House.

Ehrlich, P., A. Ehrlich, et al. (1977). Ecoscience: population, resources, environment. . San Francisco, W.H. Freeman.

Farber, S. and R. Costanza (1987). "The Economic Value of Wetlands Systems." Journal of Environmental Management **24**(1): 41-51.

Farber, S., R. Costanza, et al. (2006). "Linking ecology and economics for ecosystem management." Bioscience **56**(2): 121-133.

Farber, S., R. Costanza, et al. (2002). "Economic and Ecological Concepts for Valuing Ecosystem Services." Ecological Economics **41**(3): 375-392.

Farber, S. and B. Griner (2000). "Using conjoint analysis to value ecosystem change." Environmental Science & Technology **34**(8): 1407-1412.

Flores, N. E. and J. Thacher (2002). "Money, who needs it? Natural resource damage assessment." Contemporary Economic Policy **20**(2): 171-178.

Freeman III, A. K. (2003). The measurement of environmental and resources values. Washington, D,C,, Resource for the Future.

Froehlich, M., D. J. C. Hufford, et al. (1991). The United States. Valuing the environment: six case studies. J.-P. Barde and D. W. Pearce. London, Earthscan Publications

Limited.

Georgescu-Roegen, N. (1971). The entropy law and the economic process. Cambridge, MA, Harvard University Press.

Gren, I.-M. (2003). Monetary green accounting and ecosystem services. Working papre No.86, July 2003. Stockholm, The National Institute of Economic Research.

Gret-Regamey, A. and S. Kytzia (2007). "Integrating the valuation of ecosystem services into the Input-Output economics of an Alpine region." Ecological Economics **63**(4): 786-798.

Hahn, R. W. (2000). "The impact of economics on environmental policy." Journal of Environmental Economics and Management **39**(3): 375-399.

Hanemann, M. (1992). Preface. Pricing the European environment. S. Navrud. Oslo, Scandinavian University Press**:** 9-35.

Hanemann, W. M. (1994). "Valuing the Environment through Contingent Valuation." Journal of Economic Perspectives **8**(4): 19-43.

Hanley, N., D. MacMillan, et al. (1998). "Contingent valuation versus choice experiments: Estimating the benefits of environmentally sensitive areas in Scotland." Journal of Agricultural Economics **49**(1): 1-15.

Hanley, N. and C. Spash (1993). Cost-Benefit Analysis and the Environment. Cornwall, Edward Elgar Publishing.

Hausman, J. A. (1993). Contingent valuation: a critical assessment. Amsterdam, North-Holland.

Higgins, S. I., J. K. Turpie, et al. (1997). "An ecological economic simulation model of mountain fynbos ecosystems - Dynamics, valuation and management." Ecological

Economics **22**(2): 155-169.

Hotelling, H. (1949). Letter to the Director of the National Park Service. The economics of public recreation: the Prewitt report. R. A. Prewitt. Washington, D.C., Department of Interior.

Howarth, R. B. and S. Farber (2002). "Accounting for the value of ecosystem services." Ecological Economics **41**(3): 421-429.

Howarth, R. B. and M. A. Wilson (2006). "A Theoretical Approach to Deliberative Valuation: Aggregation by Mutual Consent." Land Economics **82**: 1-16.

Hwang, C. L. and K. Yoon (1981). Multiple Attribute Decision Making: Methods and Applications – A State of the Art Survey. New York, Spriner-Verlag.

Jacobs, M. (1997). Environmental valuation, deliberative democracy and public decision-making. Valuing Nature: Economics, Ethics and Environment,. J. Foster. London, Rutledge**:** 211-231.

Just, R. E., L. H. Darrell, et al. (1982). "Applied welfare economics and public policy." American Economic Review **69**(5): 947-954.

Just, R. E. and D. L. Heuth (1979). Applied welfare economics and public policy. Englewood Cliffs, NJ, Prentice-Hall.

Kahneman, D. and J. L. Knetsch (1992). "Valuing Public-Goods - the Purchase of Moral Satisfaction." Journal of Environmental Economics and Management **22**(1): 57-70.

Krutilla, J. V. (1967). "Conservation reconsidered." The American Economic Review **57**(4): 777-789.

Loomis, J. (1999). Contingent valuation methodology and the US institutional framework. Valuing environmental preferences. I. Bateman and M. Williams. New York, Oxford University Press**:** 613-629.

Loomis, J. B. and D. S. White (1996). "Economic benefits of rare and endangered species: Summary and meta-analysis." Ecological Economics **18**(3): 197-206.

Mackenzie, J. (1992). "Evaluating Recreation Trip Attributes and Travel Time Via Conjoint-Analysis." Journal of Leisure Research **24**(2): 171-184.

Martinez-Alier, J., G. Munda, et al. (1998). "Weak comparability of values as a foundation for ecological economics." Ecological Economics **26**(3): 277-286.

Matero, J. and O. Saastamoinen (2007). "In search of marginal environmental valuations - ecosystem services in Finnish forest accounting." Ecological Economics **61**(1): 101-114.

McCauley, D. J. (2006). "Selling out on nature." Nature **443**(7107): 27-28.

McCollum, D. W. (2003). Nonmarket valuation in action. A primer on nonmarket valuation. P. A. Champ, K. J. Boyle and T. C. Brown. Dordrecht, The Netherlands, Kluwer Academic Publishers. **483-535**.

McConnell, K. E. and N. Bockstael (2006). Valuing the Environment as a Factor of Production Handbook of Environmental Economics, Vol 2. . K. G. Maler and J. Vincent, Elsevier**:** 621-669.

Meadows, D. H., D. L. Meadows, et al. (1972). The limits to growth. New York, Potomac Associates.

Millennium, E. A. (2003). Ecosystems and Human Well-Being: A Framework for Assessment. Washington DC., Island Press.

Mitchell, R. C. and R. T. Carson (1989). Using surveys to value public goods: the contingent valuation method. Baltimore, John Hopkins University Press.

Munda, G. (1995). Multicriteria evaluation in a fuzzy environment, Physica-Verlag.

Navrud, S. and G. J. Pruckner (1997). "Environmental valuation -- to use or not to use? a comparative study of the United States and Europe." Environmental & Resource Economics **10**(1): 1-26.

Norgaard, R. B. and C. Bode (1998). "Next, the value of God, and other reactions." Ecological Economics **25**(1): 37-39.

Odum, E. P. (1979). "Rebuttal of ''economic value of natural coastal wetlands: a critique''." Coastal Zone Manage **J 5**: 231-237.

Odum, H. T. (1967). Energetics of world food production. In: Problems of world food supply. President's Science Advisory Committee Report. Volume 3. Washington, DC, The White House**:** 55–94.

Odum, H. T. (1971). Environment, power, and society. New York, Wiley-Interscience.

Odum, H. T. (1979). "Principle of environmental energy matching for estimating potential value: a rebuttal." Coastal Zone Manage **J 5**: 239-241.

Office of Policy Analysis and Environmental Protection Agency (1987). EPA's use of cost benefit analysis: 1981-1986.

Olschewski, R., T. Tscharntke, et al. (2006). "Economic evaluation of pollination services comparing coffee landscapes in Ecuador and Indonesia." Ecology and Society **11**(1).

Pearce, D. W. and T. Seccombe-Hett (2000). "Economic valuation and environmental decision-making in Europe." Environmental Science & Technology **34**(8): 1419-1425.

Perrings, C. (2006). "Ecological Economics after the Millennium Assessment."

International Journal of Ecological Economics & Statistics **6**: 8-23.

Prato, T. (1999). "Multiple attribute decision analysis for ecosystem management." Ecological Economics **30**(2): 207-222.

Priess, J. A., M. Mimler, et al. (2007). "Linking deforestation scenarios to pollination services and economic returns in coffee agroforestry systems." Ecological Applications **17**(2): 407-417.

Ricketts, T. H., G. C. Daily, et al. (2004). "Economic value of tropical forest to coffee production." Proceedings of the National Academy of Sciences of the United States of America **101**(34): 12579-12582.

Ridker, R. G. and J. A. Henning (1967). "the determinants of residential property values with special reference to air pollution." The Review of Economics and Statistics **49**(2): 246-257.

Ropke, I. (2004). "The early history of modern ecological economics." Ecological Economics **50**: 293-314.

Rosenberger, R. S. and J. B. Loomis (2000). "Using meta-analysis for benefit transfer: In-sample convergent validity tests of an outdoor recreation database." Water Resources Research **36**(4): 1097-1107.

Schmalensee, R. (1976). "Another look at social valuation of input price changes." American Economic Review **66**(1): 239-243.

Shabman, L. A. and S. S. Batie (1978). "Economic value of natural coastal wetlands: a critique." Coastal Zone Manage **J 4**(3): 231-247.

Shrestha, R. K. and J. B. Loomis (2003). "Meta-analytic benefit transfer of outdoor recreation economic values: Testing out-of-sample convergent validity."

Environmental & Resource Economics **25**(1): 79-100.

Silva, P. and S. Pagiola (2003). A review of the valuation of environmental costs and benefits in World Bank projects. Washington, D.C.

Smith, V. K. (1984). Environmental policy making under executive order 12291: an introduction. Environmental policy under Reagan's executive order. V. K. Smith, The University of North Carolina Press**:** 3-40.

Smith, V. K. (1993). "Nonmarket Valuation of Environmental Resources - an Interpretive Appraisal." Land Economics **69**(1): 1-26.

Smith, V. K. (2000). "JEEM and non-market valuation: 1974-1998." Journal of Environmental Economics and Management **39**(3): 351-374.

Smith, V. K. and Y. Kaoru (1990). "What Have We Learned since Hotelling Letter - a Meta-analysis." Economics Letters **32**(3): 267-272.

Troy, A. and M. A. Wilson (2006). "Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer." Ecological Economics **60**(2): 435-449.

Turner, R. K., J. Paavola, et al. (2003). "Valuing nature: lessons learned and future research directions." Ecological Economics **46**(3): 493-510.

US National Research Council (2005). Valuing Ecosystem Services: Toward Better environmental Decision-Making. Washington, D.C., The National Academies Press.

Vatn, A. and D. W. Bromley (1994). "Choices without Prices without Apologies." Journal of Environmental Economics and Management **26**(2): 129-148.

Walsh, R. G., D. M. Johnson, et al. (1989). "Issues in nonmarket valuation and policy application: a retrospective glance." Western Journal of Agricultural Economics **14**: 178-188.

Walsh, R. G., D. M. Johnson, et al. (1992). "Benefit Transfer of Outdoor Recreation Demand Studies, 1968-1988." Water Resources Research **28**(3): 707-713.

Weisbrod, B. A. (1964). "Collective consumption services of individual consumption goods." Quarterly Journal of Economics **77**(Aug.): 71-77.

Wilson, M. and A. Troy (2003). Accounting for the economic value of ecosystem services in Massachusetts. Losing Ground: at what cost. K. Breunig. Boston, Massachusetts Audubon Society.

Wilson, M., A. Troy, et al. (2004). The Economic Geography of Ecosystem Goods and Services:Revealing the monetary value of landscapes through transfer methods and Geographic Information Systems. Cultural Landscapes and Land Use. M. Dietrich and V. D. Straaten, Kluwer Academic.

Wilson, M. A. and J. P. Hoehn (2006). "Valuing environmental goods and services using benefit transfer: The state-of-the art and science." Ecological Economics **60**(2): 335-342.

Wilson, M. A. and R. B. Howarth (2002). "Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation." Ecological Economics **41**(3): 431-443.

Woodward, R. T. and Y. S. Wui (2001). "The economic value of wetland services: a meta-analysis." Ecological Economics **37**(257-270).

Valuing New Jersey's Ecosystem Services and Natural Capital: A Benefit Transfer Approach*

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ABSTRACT 94 peer-reviewed environmental economic studies were used to value ecosystem services in the State of New Jersey. The benefit estimate was translated into 2004 US dollars per acre per year, we then computed the average value for a given eco-service for a given ecosystem, and multiplied the average by the total statewide acreage for that ecosystem. The total value of these ecosystem services is $11.6 billion/year and we believe that these estimates are almost certainly conservative. The result from this value transfer exercise is a useful, albeit imperfect, basis for assessing and comparing these services with the value of conventional economic goods and services.

KEY WORDS: Ecosystem service valuation; Natural capital; Ecosystem management; Trade-offs; Benefit transfer

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* The methodology and result sections of this paper were adapted from Costanza et al. (2007).

Natural capital consists of those components of the natural environment that provide a long-term stream of benefits to individual people and to society as a whole. The benefits provided by natural capital include both goods and services; goods come from both ecosystems (e.g., timber) and abiotic (non-living) sources (e.g., mineral deposits), while services are mainly provided by ecosystems. Examples of ecosystem services include temporary storage of floodwaters by wetlands, long-term storage of climate-altering greenhouse gases in forests, dilution and assimilation of wastes by rivers, and numerous others. All of these services provide economic value to people.

For policy, planning, and regulatory decisions, it is important for New Jersey residents to know not only what ecosystem goods and services will be affected by public and private actions, but also what their economic value is relative to other market and non-market goods and services, such as those provided by physical capital (e.g., roads), and human capital investment (e.g., education), etc.

Of course, it may be very difficult (given our present knowledge) to assign a defensible value to some aspects of the environment. While the benefits of environmental preservation and the environmental costs of development are familiar, they are often not treated in economic terms in the same sense as, say, the cost of a new school or highway. In part this omission stems from the fact that the impacts on the natural environment are often difficult to quantify in physical and monetary terms, which makes it hard to know exactly what we are gaining when we preserve a landscape in its undeveloped state or what we lose when we decide not to protect a natural area.

To address this inadequacy, citizens, business leaders and government decision makers need to know whether the benefits of development postulated by its supporters—

jobs, income, and tax revenues–will be overshadowed by unseen costs in the future. The challenge, in short, is to make the linkages between landscape and the human values it represents as explicit and transparent as possible. The identification and measurement of environmental features of value is also essential for the efficient and rational allocation of environmental "resources" among competing demands on natural and cultural landscapes (Daily 1997, Costanza et al. 1997, Wilson and Carpenter 1999).

This study aims to present an assessment of the economic benefits provided by New Jersey's natural environment by using benefit transfer to generate value estimates that can be integrated into land use planning and environmental decision-making throughout the state.

BACKGROUND AND METHODS

Ecosystem services and valuation (ESV)

Benefits associated with the natural environment are often described in terms of "natural resources", including both non-living resources such as mineral deposits and living resources such as timber, fertile soil, fish, etc. The emphasis in this conceptual framework is on things of value that can be extracted from the environment for direct use by humans. A different way of looking at environmental benefits has been gaining favor over the last several decades. In this "natural capital" or "ecosystem services" framework, the natural environment is viewed as a "capital asset", i.e., an asset that provides a flow of benefits over an extended period (Costanza and Daly 1992). While non-living resources are not ignored, the emphasis is on the benefits provided by the living environment, usually viewed in terms of a whole *ecosystem*, which is defined as all the interacting abiotic and biotic elements of an area of land or water. *Ecosystem functions* are the processes of transformation of matter and energy in ecosystems. *Ecosystem goods and services* are the benefits that humans derive (directly and indirectly) from naturally functioning ecological systems (Costanza et al. 1997, Daily 1997, De Groot et al. 2002, Wilson et al. 2004, Millennium Ecosystem Assessment 2003).

In addition to the production of marketable goods, ecosystems provide natural functions such as nutrient recycling as well as conferring aesthetic benefits to humans. Ecosystem goods and services may therefore be divided into two general categories: *market goods and services* and *non-marke goods and services.* While measuring market values simply requires monitoring market data for observable trades, non-market values of goods and services are much more difficult to measure. When there are no explicit markets for services, a more indirect means of assessing values must be used. A spectrum of valuation techniques commonly used to establish values when market values do not exist has been developed (Freeman 2003, Champ et al. 2003, cf. Farber et al. 2006 for a brief review).

Benefit transfer

Benefit transfer is defined as the adaptation of existing ESV information or data to new policy contexts which have little or no data. The transfer method involves obtaining an estimate for the value of ecosystem services through the analysis of a single study, or group of studies, that have been previously carried out to value “similar” goods or services in “similar” locations. The transfer itself refers to the application of derived values and other information from the original ‘study site’ to a ‘policy site’ which can vary across geographic space and/or time (Brookshire and Neill 1992, Desvousges et al. 1992). For example, an estimate of the benefit obtained by tourists viewing wildlife in one park (study site) might be used to estimate the benefit obtained from viewing wildlife in a different park (policy site).

Over time, the transfer method has become a practical way of making informed decisions when primary data collection is not feasible due to budget and time constraints (Moran 1999). Primary valuation research is always a "first-best" strategy in which information is gathered that is specific to the location and action being evaluated. However, when primary research is not possible or plausible, then benefit transfer, as a "second-best" strategy, is important to evaluating management and policy impacts. For instance, EPA's regulation development process almost always involves benefit transfer. Although it is explicitly recognized in the EPA's *Guidelines for Preparing Economic Analyses (2000)* that this is not the optimal situation, conducting an original study for anything but the most significant policies is almost impossible. This is due to the fact that any primary research must be peer-reviewed if it is to be accepted for regulation development, which requires both time and money (Griffiths 2002).

Of course, the quality of the original studies used in the benefit transfer exercise always determines the overall quality and scope of the final value estimate (Brouwer 2000). In this study we were able to identify three categories of valuation research[1] and only focused on Type A studies, which include peer-reviewed empirical analyses using

[1] Type B studies are commonly referred to as 'grey literature' and generally represent non peer-reviewed analyses such as technical reports, PhD Theses and government documents using conventional environmental economic techniques that also focus on individual consumer preferences. Type C studies represent secondary, summary studies such as statistical meta-analyses of primary valuation literature which include both conventional environmental economic techniques as well as non-conventional techniques (Energy analyses, Marginal product estimation) to generate synthesis estimates of ecosystem service values.

conventional environmental economic techniques (e.g., Travel Cost, Hedonic Pricing and Contingent Valuation) to elicit individual consumer preferences for environmental services.

In addition to being peer-reviewed, a study also has to satisfy two criteria to be selected: 1) its research area has to be temperate regions in North America and Europe to ensure similarity between the study site and the transfer site, and 2) it has to focus primarily on non-consumptive use.

A total of 94 studies covering the types of ecosystems present in New Jersey were identified for benefit transfer. Because some studies provided more than one estimated ecosystem service value for a given ecosystem; the set of 94 studies provided a total of 163 individual value estimates. We translated each estimate into dollars per acre per year, computed the average value for a given ecosystem service for a given ecosystem, and multiplied the average by the total statewide acreage for that ecosystem generated from Geographical Information Systems (GIS). The following formula is used in calculating total ecosystem services:

$$V(ESV_i) = \sum_{i=1}^{n} A(LU_i) \times V(ES_{ki})$$

Where A(LUi) = Area of Land Use (i) and

V(ESVi) = Annual value of Ecosystem Services (k) for each Land Use (i)

Spatially-explicit benefit transfer

Geographical Information Systems (GIS) have been used to increase the context specificity of value transfer (e.g. Eade and Moran 1996, Wilson et al. 2004, Troy and Wilson, 2006). In doing so, the value transfer process is augmented with a set of spatially explicit factors so that geographical similarities between the policy site and the study site are more easily detected. In addition, the ability to present and calibrate economic valuation data in map form offers a powerful means for expressing environmental and economic information at multiple scales to stakeholders.

In simplified terms, the technique involves combining one land cover layer with another layer representing the geography by which ecosystem services are aggregated - i.e. watershed, town or park.

A New Jersey-specific land cover typology was developed by the research team for the purposes of calculating and spatially assigning ecosystem service values. This typology is a variant of the New Jersey Department of Environmental Protection (NJDEP) classification for the 1995/97 Land use/Land cover (LULC) by Watershed Management Area layer.[2] The new typology condenses a number of DEP classes having similar (or no) ecosystem service value and creates several new classes to reflect important differences in ecosystem service values that occur within a given DEP class. The development of the land cover typology began with a preliminary survey of available GIS data for New Jersey to determine the basic land cover types present and the level of categorical precision in those characterizations. This process resulted in a unique 13-class land cover typology for the State of New Jersey.

[Insert Table 1]

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[2] At the time the research for this report was conducted, 1995/1997 land use/land cover data was the most recent available.

To date, there are only a limited number of published analyses using a spatial value transfer framework (cf. Troy and Wilson 2006 for a brief review) and we are not aware of any done at the state level.

RESULTS

Gap analysis

Part of the value of going through an ecosystem services evaluation is to identify the gaps in existing information to show what types of research are needed. The data reported in the light grey boxes in Table 1 show 163 individual ESV estimates obtained from 94 individual peer-reviewed empirical valuation papers on the land cover types included in this study. Areas shaded in white represent situations where we do not anticipate a particular ecosystem service to exist in a particular land cover type (i.e., pollination in the coastal shelf). Areas shaded in dark grey represent cells where we do anticipate a service to exist or be provided by a land cover type, but for which there is currently no empirical research available that satisfies our search criteria.

This "gap analysis" indicated that not all land cover types could be effectively matched with all possible ecosystem services for each individual land cover type in the State of New Jersey. Only 26% of the cells are filled.

This is partially because the research team's search criteria were focused primarily on Type A economic valuation results. But more importantly, many landscapes that are of interest from an environmental management perspective simply have not yet been studied for their non-market ecosystem service values.

The valuation of ecosystem services is an evolving field of study and to date it has not generally been driven by ecological science or policy needs; instead it has been guided primarily by economic theory and methodological constraints. Therefore, we expect that as the field continues to mature, landscape features of interest from an ecological or land management perspective in New Jersey will increasingly be matched up to economic value estimates. As more primary empirical research is gathered, we anticipate that higher, not lower, aggregate values will be forthcoming for many of the land cover types represented in this study. This is because, as discussed above, several ecosystem services that we might reasonably expect to be delivered by healthy, functioning forests, wetlands and riparian buffers simply remain unaccounted for in the present analysis. As more of these services are better accounted for, the *total* estimated value associated with each land cover type will likewise increase.

[Insert Table 2]

Per unit value of ecosystem services

Using the list of land cover classes shown in Table 1, queries were conducted of the best available economic valuation data to generate baseline ecosystem service values estimates for the entire study area in New Jersey. All results were standardized to average 2004 U.S. dollar equivalents per acre/per year to provide a consistent basis for comparison below. The aggregated baseline ESV results for all land cover types represented within the study area are presented below in Table 3.

[Insert Table 3]

Each cell presents the standardized average ESV for ecosystem services associated with each of the unique land cover types. For purposes of clarity and in line with recent practice (e.g. Costanza et. al. 1997, Eade and Moran 1999) all results represent the statistical mean for each land cover/ecosystem service pairing unless otherwise specified. Because each average value can be based on more than one estimate, the actual number of estimates used to derive each average ecosystem service value is reported separately in Appendix A and detailed information for the literature sources used to calculate estimates for each ecosystem service-land cover pair is available upon request.

Moreover, for purposes of transparency, in addition to presenting a single point estimate for each land cover/ecosystem service pair, the minimum, maximum, and median dollar values are also presented for further review in Appendix A at the end of this dissertation. As these tables reveal, means do tend to be more sensitive to upper bound and lower bound outliers in the literature, and therefore some differences do exist between the mean and median estimates. For example, the mean for beach ESV is approximately forty two thousand dollars per acre per year, while the median is thirty eight thousand, a difference of approximately four thousand dollars per year. Given that a difference of approximately four thousand dollars represents the largest mean-median gap in our analysis, however, we are confident that the results reported here would not dramatically change if means were replaced with medians[3].

[3] While it may also be tempting to narrow statistical ranges by discarding high and low 'outliers' from the literature, the data used was directly derived from empirical studies rather than theoretical models and there is no defensible reason for favoring one set of estimates over another. Data trimming therefore was not used.

The valuation results in Table 3 were generated from 94 unique Type A studies collected by the research team. As the summary column at the far right of the table shows, there is considerable variability in ecosystem service values delivered by different land cover types in New Jersey. As expected, the data in the table reveals that there is a fairly robust spread of ESVs delivered by different land cover types, with each land cover representing a unique mix of services documented in the peer-reviewed literature. On a per acre basis, for example, beaches appear to provide the highest annual ESV flow values for the State of New Jersey ($42,147) with disturbance control ($27,276) and aesthetic/recreation values ($14,847) providing the largest individual values to that aggregated sum respectively[4]. Next, it appears that both freshwater wetlands ($8,695) and saltwater wetlands ($6,527) contribute significantly to the annual ESV flow throughout the State of New Jersey. On the lower end of the value spectrum, cropland ($23) and grassland/rangeland ($12) provide the lowest annual ESV flow values on an annualized basis. While significantly different from the other land cover types, this finding is consistent with the focus of the current analysis on *non-market* values, which by definition exclude provisioning services provided by agricultural landscapes (i.e. food and fodder).

The column totals at the bottom of Table 3 also reveal considerable variability between the averages ESVs delivered by different ecosystem service *types* in New Jersey. Once each average ESV is multiplied by the area of land cover type which provides it, and is summed across all possible combinations, both water regulation and aesthetic/recreational services clearly stand out as the largest ecosystem service

[4] This finding is consistent with the Hedonic regression analysis presented in this report.

contributors in New Jersey, cumulatively representing over $6 billion in annual value. At the other end of the spectrum, due to gaps in the peer-reviewed literature, soil formation, biological control, and nutrient cycling appear to contribute the least value to New Jersey.

Once the annualized dollar value *per acre* was identified, ecosystem service flow values for land cover types in New Jersey were determined by multiplying the areas of each cover type, in acres, by the *per acre* estimate for that cover type. These results are summarized below in Table 4. The estimates were then mapped by HUC 14 subwatersheds across the state of New Jersey. This was done by combining DEP's watershed management area layer with the modified LULC layer. The results of the operation included the area and the land cover type for each subwatershed. Maps were then created using a graduated color classification to show both per acre and total ESV estimates for all New Jersey subwatersheds.

Here, the data clearly shows that substantial economic value is delivered to New Jersey citizens every year by functioning ecological systems in the landscape. The total value of ecosystem services is approximately ***$11 billion per year*** (Table 4). Consistent with the value transfer data reported above in Table 3, it appears that ecosystem services associated with both freshwater and saltwater wetland types, as well as forests and estuaries, tend to provide the largest cumulative economic value.

[Insert Table 4]

As the following maps of New Jersey show (Figures 1-2), there is considerable heterogeneity in the actual delivery of ESV's across the New Jersey landscape with particularly notable differences between interior and coastal watersheds across the state. For example, on close examination, as expected, it appears that watersheds associated with an abundance of freshwater wetlands consistently reveal the highest ESV flow values statewide.

[Insert Figure 1 and Figure 2]

Net present value of natural capital and sensitivity analysis

If we think of ecosystem services as a stream of annual “income”, then the ecosystems that provide those services can be thought of as part of New Jersey’s total *natural capital*. To quantify the value of that capital, we must convert the stream of benefits from the future flows of ecosystem services into a net present value (NPV). This conversion requires some form of discounting. Discounting of the flow of services from natural assets is somewhat controversial (Azar and Sterner 1996. For a recent debate on the choice of a discount rate on climate change see Nordhaus 2007 *vs* Stern and Taylor 2007). The simplest case involves assuming a constant flow of services into the indefinite future and a constant discount rate. Under these special conditions, the NPV of the asset is the value of the annual flow divided by the discount rate.

The discount rate one chooses here is a matter of debate. Previous work (i.e. Costanza et al. 1989) indicated a major source of uncertainty in the analysis is the choice of discount rate. Beyond this, there is also some debate over whether one should use a zero discount rate or whether one should even assume a constant discount rate over time. A constant rate assumes “exponential” discounting, but “decreasing,” “logistic,” “intergenerational,” and other forms of discounting have also been proposed (i.e. Azar and Sterner 1996, Sumaila and Walters, 2005, Weitzman 1998, Newell and Pizer 2003).

Table 5 shows the results using a range of constant discount rates along with other approaches to discounting, including using a decreasing discount rate, intergenerational discounting, and 0% discounting using a limited time frame. The general form for calculating the NPV is:

$$NPV = \sum_{t=0}^{\infty} V_t W_t$$

Where:

V_t = the value of the service at time t

W_t = the weight used to discount the service at time t

For standard exponential discounting, W_t is exponentially decreasing into the future at the discount rate, r.

$$W_t = \left(\frac{1}{1+r}\right)^t$$

Applying this formula to the annual ecosystem service flow estimates of $10 billion per year for a range of discount rates (r) from 0% to 8% yields the first row of estimates in Table 5. Note that for a 0% discount rate, the value of equation 1 would be infinite, so one needs to put a time limit on the summation. In Table 5, we assumed a 100 year time frame for this purpose, but one can easily see the effects of extending this time frame. An annual ecosystem service value of $11 Billion for 100 years at a 0% discount rate yields an NPV of $1.1 trillion. This estimate turns out to be identical to the NPV calculated using a 1% discount rate and an infinite time frame. As the discount rate increases, the NPV decreases. At an 8% discount rate an annual flow of $11 billion translates to an NPV of $138 billion.

[Insert Table 5]

Another general approach to discounting argues that discount rates should not be constant, but should decline over time. There are two lines of argument supporting this conclusion. The first, according to Weitzman (1998) and Newell and Pizer (2003), argues that discount rates are uncertain and because of this their average value should decline over time. As Newell and Pizer (2003, pp. 55) put it: "future rates decline in our model because of dynamic uncertainty about future events, not static disagreement over the correct rate, nor an underlying belief or preference for deterministic declines in the discount rate." A second line of reasoning for declining rates is attributed to Azar and Sterner (1996), who first decompose the discount rate into a "pure time preference" component and an "economic growth" component. Those authors argue that, in terms of social policy, the pure time preference component should be set to 0%. The economic growth component is then set equal to the overall rate of growth of the economy, under the assumption that in more rapidly growing economies there will be more income in the future and its impact on welfare will be marginally less, due to the assumption of decreasing marginal utility of income in a wealthier future society. If the economy is assumed to be growing at a constant rate into the indefinite future, this reduces to the standard approach of discounting, using the growth rate for r. If, however, one assumes that there are fundamental limits to economic growth, or if one simply wishes to incorporate uncertainty and be more conservative about this assumption, one can allow the assumed growth rate (and discount rate) to decline in the future.

As an example, (following Newell and Pizer 2003, who based their rates of decline on historical trends in the discount rate), we let the discount rate approach 0 as time approaches 300 years into the future. This is done by multiplying r by e^{-kt}, where k was set to .00007. Because this function levels out at a discount rate of 0%, W_t eventually starts to increase again. W_t is therefore forced to level out at its minimum value. Also, carrying this calculation to infinity would lead to an infinite NPV. For this example, the summation was carried out for 300 years (which is the time frame used by Newell and Pizer (2003). As one can see from an inspection of Table 5, in general, assuming a decreasing discount rate leads to significantly higher NPV values than assuming a constant discount rate.

Finally, we applied a recently developed technique called "intergenerational discounting" (Sumaila and Walters 2005). This approach includes conventional exponential discounting for the current generation, but it also includes conventional exponential discounting for future generations. Future generations can then be assigned separate discount rates that may differ from those assumed for the current generation. For the simplest case where the discount rates for current and future generations are the same, this reduces to the following formula (Sumaila and Walters 2005, pp. 139):

$$W_t = d^t + \frac{d * d^{t-1} * t}{G}$$

Where:

$$d = \frac{1}{1+r}$$

G = the generation time in years (25 for this example)

One can see that this method leads to significantly larger estimates of NPV than standard constant exponential discounting, especially at lower discount rates. At 1% the NPVs are 5 times as great, while at 3% they are more than twice as large.

Any choice of discount rate and discounting approach is a matter of both the empirical and ethical (Tol 1999). It is empirical because people make trade-offs between the present and the future in their economic decisions. It is ethical because the discount rate determines the allocation of intertemporal goods and services between generations.

Newell and Pizer (2003) argue for a 4% discount rate, declining to approximately 0% in 300 years, based on historical data. One could argue that for ecosystem services the starting rate should be lower (e.g. Stern used a utility discount rate of 0.001 and a consumption discount rate of 0.014 in his recent report on the economics of climate change). If we use 3% and focus on the two alternative methods, this would place the NPV of New Jersey's natural capital assets at around $0.6 trillion.

DISCUSSION

Validity and reliability of the transfer result

The *validity* of a measure is the degree to which it measures the theoretical construct under investigation. Reliability is the "consistency" or "repeatability" of measures. A measure is considered reliable if it would give us the same result over and over again. We will discuss below the validity and reliability of our benefit transfer result.

Convergent validity test

Benefit transfer estimates are of great interest to practitioners, provided that they can be proven to be adequate surrogates for on-site estimates achievable by conducting costly original studies. While the practical allure is clear, can benefit transfer provide reasonable and meaningful estimates of ecosystem service value? The scientific issue here can be framed in terms of the concept of theoretical *validity*, which has been explained by Mitchell and Carson (1989, p. 190):

> "The *validity* of a measure is the degree to which it measures the theoretical construct under investigation. This construct is, in the nature of things, unobservable; all we can do is to obtain imperfect measures of that entity" (Italics added).

In the context of benefit transfer, the "theoretical construct under investigation" is an estimate that has been derived from an original study site. The true value itself is unobservable (i.e., it is cannot be measured directly) so the user has no way of determining its "real" value. All the analyst can do is to try to make the transferred value -- an imperfect surrogate of the "real" value – acceptable or *valid* for transfer.

So, the question arises: how does the policy maker know when the transferred value is valid or not if there is no "real" value to compare it with? One answer is to introduce another estimated value of the item as a baseline for comparison--which is in many cases obtained from an original study—and see if it is *convergent* with the transferred value. The two value estimates are then compared and if they are not statistically different, *convergent validity* of value transfer is established (Bishop et al. 1997).

In this study we compared our transferred results with those derived from a Hedonic Pricing (HP) study to see whether the convergent validity criterion is met. Hedonic analysis is one method that can be used to estimate the amenity value of ecosystems. This approach statistically separates the effect on property values of proximity to environmental amenities (such as protected open space or scenic views) from other factors that affect housing prices.

In this specific HP study, the study site consisted of seven local housing markets located in Middlesex, Monmouth, Mercer and Ocean Counties of the State of New Jersey. In most respects those markets are demographically similar in the aggregate to the state as a whole (cf. Costanza et al. 2007 for technical details). The results demonstrate that homes that are closer to environmental urban green space and beaches generally sell for more than homes further away, all else being equal. The benefit estimates were similar to those derived from the benefit transfer approach but were considerably higher. For urban greenspace the annual value ranged from $10,015 to $11,066 per acre (using a 3% discount rate) compared to the $2473 derived from the benefit transfer. In the case of beaches, the value range is between $31,540 to $43,718 compared to the benefit transfer estimate of $42,147.

Standard deviations as a measure of reliability

Table 6 presents the standard deviation (SD) of the means for different value estimates within and across studies for each ecosystem service/land cover pairing. The first and second number in the parentheses indicates the number of studies and observations from which the SD calculated, respectively.

10 of the 35 filled cells are based on a single observation (and therefore have a zero standard deviation). Three estimates are based on a single study that in each case provides more than one observation. Where transferred results are based on more than one study the standard deviation is larger than the mean in around half the cases.

How to explain these large variances? There are three possible sources: 1) generalization errors 2) measurement error related to original research, and 3) measurement error in the benefit transfer process. Next we will discuss each potential source in detail.

[Insert Table 6]

Possible Sources of Error

Generalization errors

Benefit transfer assumes that there is an underlying meta-valuation function so that variance in ecosystem services value could be explained by biophysical and socio-economics attributes across time and space. Generalization errors occur when estimates from study sites are adapted to represent different policy sites. These errors are inversely related to the degree of similarity between the two sites (Rosenberger and Stanley 2006).

Because developing a meta-function was not possible due to time and budget constraints, point transfer was used in this study. Ideally value estimates from the primary studies are random draws and therefore are normally distributed and their average will be a close approximation of the population mean. However, this is not the case for a couple of reasons.

First, the primary studies were not randomly selected. Only peer-reviewed literature was included because of its presumably higher overall quality. However, these value estimates might be systematically higher or lower compared to non-peer-reviewed sources. Several recent meta-analyses explicitly model the effect of publication source and results are mixed depending on the methodology applied and the commodity valued (Rosenberger and Stanley 2006).

Second, only valuation studies with study areas in North American and European countries are included. This is because we expect there is a similarity in socio-economic factors (income, and attitude towards the environment, etc) between these areas and New Jersey that could reduce generalization errors.

These socio-economic factors together with land cover type and the ecosystem service that is being valued are the only attributes controlled during the point transfer process. Many factors were not taken into account, such as methodology, type and degree of marginal change the value estimates were associated with, all of which have been shown to be significant in explaining the variance of value estimates by various meta-analyses. As an example, even three estimates from the same study have a standard deviation higher than the mean in Table 6 (waste treatment service provided by saltwater wetland).

Given this information one should not be surprised to see some large variances in the transferred benefit estimates as shown in Table 6. Theoretically, during the transfer process the more variables the researcher can control, the more likely the result will be valid. In this sense, meta-analysis provides a more robust transfer because it attempts to statistically measure systematic relationships between valuation estimates and these contextual attributes (Loomis 1992).

In order to minimize the generalization error, we did not trim our data. The 94 studies we analyzed encompass a wide variety of time periods, geographic areas, investigators, and analytic methods. The present study preserves this variance; no studies were removed from the database because their estimated values were thought to be “too high” or “too low” and limited sensitivity analyses were performed.

Measurement error related to original research

Measurement error arises when researchers' decisions affect the accuracy of the benefit transfer (Rosenberger and Stanley 2006). For example, in the context of a primary study these decisions include how to phrase a survey question so that it is less likely to cause bias in responses and whether to delete outliers. During the process of benefit transfer, researchers have to make their own judgment on which primary data to include, how to aggregate the result, etc. We will first discuss the measurement errors associated with original studies.

The quality of original studies used in the benefit transfer exercise always determines the overall quality and scope of the final value estimate (Brouwer 2000). As Brookshire and Neill put it (1992), "Benefit transfers can only be as accurate as the initial benefit estimates." For the sake of quality control we elected to only consider peer-reviewed literature in our analysis. No further step was taken to decide which papers were of better quality than others because there is no quality indicator available to compare studies using different methods.

Of course there are a couple of assumptions involved by choosing the peer-reviewed studies only: first, they are of higher quality, and second, the higher the quality, the more accurately "true" value is measured and measurement errors minimized. Another type of measurement error related to original studies has nothing to do with the quality of the individual studies but is due to the limited number and scope of the available studies. This too, will inevitably affect the benefit transfer process. Here are a couple of examples:

As the gap analysis shows, incomplete coverage is a serious issue. Not all ecosystems have been well studied and some have not been studied at all. More complete coverage of ecosystem services would almost certainly increase the aggregate values shown in Table 4. In our project report we did include several non-peer-reviewed studies to fill in some gaps. As a result the total annual ecosystem services value in New Jersey was estimated at 19.4 billion$/year instead of the 11.6 billion$/year reported in this paper.

Most estimates are based on current willingness-to-pay or proxies, which are limited by people's perceptions and knowledge base. Improving people's knowledge base about the contributions of ecosystem services to their welfare would almost certainly increase the values based on willingness-to-pay, as people would realize that ecosystems provided more services than they had previously been aware of.

Measurement errors related to benefit transfer process

In our study the value of a non-marketed ecosystem service was obtained by multiplying the level of each service by a shadow price which represents the marginal value of that service in question. This technique is analogous to that used in calculating gross domestic product (GDP) which measures the total value of market goods and services (Howarth and Farber 2002).

However, several problems arise when one attempts to use the shadow price derived from a partial equilibrium framework in a general equilibrium context, where the changes involved are not marginal anymore. First, a static, partial equilibrium framework ignores interdependencies and dynamics. For instance, our approach probably underestimates shifts in the corresponding demand curves as the sources of ecosystem services become more limited. Second, it assumes smooth responses to changes in ecosystem quantity with no thresholds or discontinuities. Third, it assumes spatial homogeneity of services within ecosystems. One might argue that every ecosystem is unique, and per-acre values derived from elsewhere may not be relevant to the ecosystems being studied[5]. Even within a single ecosystem, the value per acre depends on the size of the ecosystem. The marginal cost per acre is generally expected to increase as the quantity supplied decreases, and a single average value is not the same thing as a range of marginal values.

Unfortunately we have far too few data points to construct a general equilibrium model to incorporate interdependencies, dynamics and thresholds. Similarly, to solve the problem of spatial homogeneity, one has to first limit valuation to a single ecosystem in a single location and using only data developed expressly for the unique ecosystem being studied, and then repeat the process for ecosystems in other locations. For a state with the size and landscape complexity of New Jersey, this approach would preclude any valuation at the state-wide level.

Because we have no way of knowing the "true" value of various ecosystem services provided by a large geographic area like the State of New Jersey, it is difficult to estimate whether our estimated value is accurate or not and, if not, whether it is too high or too low. However, theory and past research shed some light. First, if New Jersey's ecosystem services are scarcer than assumed here, their value has been underestimated in

[5] This issue was partially addressed by the spatial modeling analysis in our project report available at http://www.nj.gov/dep/dsr/naturalcap/. The results of the spatial modeling analysis do not support an across-the-board claim that the per-acre value depends on the size of the parcel. While the claim does appear to hold for nutrient cycling and probably other services, the opposite position holds up fairly well for what ecologists call "net primary productivity" or NPP.

this study. Such reductions in “supply” appear likely as land conversion and development proceed. More elaborate systems dynamics studies of ecosystem services have shown that including interdependencies and dynamics leads to significantly higher values (Boumans et al. 2002) as changes in ecosystem service levels ripple throughout the economy. Second, the presence of thresholds or discontinuities would likely produce higher values for affected services assuming (as seems likely) that such gaps or jumps in the demand curve would move demand to higher levels than a smooth curve (Limburg et al. 2002). Third, distortions in current prices used to estimate ecosystem service values are carried through the analysis. These prices do not reflect environmental externalities and are therefore again likely to be underestimates of “true” values.

In addition to the conclusions drawn from our gap analysis and validity test, it seems most likely the “true” value of ecosystem services would involve significantly higher values. Unfortunately, it is impossible to know how much higher the values would be if these limitations were addressed. One example may be worth mentioning, however. Boumans et al. (2002) produced a dynamic global simulation model that estimated the value of global ecosystem services in a general equilibrium framework and estimated their value to be roughly twice that estimated by Costanza et al. (1997), who used a static, partial equilibrium analysis. Whether a similar result would be obtained for New Jersey is impossible to say, but it does give an indication of the potential range of values.

For future research what is needed are ESV studies that encompass ecological structures and processes, ecological functions, ecosystem services, human welfare, land use decisions and the dynamic feedback between them. To our knowledge, there have been few such studies to date (e.g. Boumans et al 2002). But it is just this type of study that is of greatest relevance to decision makers and points the way forward (Turner et al. 2003).

CONCLUSION

The total value of ecosystem services is $11.6 billion/year (USD-2004). Future flows of ecosystems services can be discounted (converted to their present value equivalents) in a number of ways; the subject of discounting is controversial and is the subject of active research, with new discounting techniques being proposed regularly. If we use conventional discounting with a constant annual discount rate of 3% (a rate often used in studies of this type), and if we assume that the $11.6 billion/yr of ecosystems services continues in perpetuity, the present value of those services, i.e. the value of the natural capital which provides the services, would be $387 billion.

We have tried to display our results in a way that allows one to appreciate the range of values and their distribution and variance (Tables 3, 6 and Appendix A). It is clear from inspection of these tables that the final estimates are not extremely precise. However, they are much better estimates than the alternative of assuming that ecosystem services have zero value, or, alternatively, of assuming they have infinite value. Pragmatically, in estimating the value of ecosystem services it seems better to be approximately right than precisely wrong.

Given the gaps in the available economic valuation data, the results presented should be treated as *conservative estimates*. In other words, the ESV results presented here are likely to underestimate, not overestimate the actual ecosystem goods and services valued by society in the State of New Jersey. As discussed previously, due to limitations of the scope and budget associated with this project, the research team was not able to include technical reports and “grey” literature in this analysis. This data gap is not unique to the present analysis and we anticipate that in the future it will be possible to expand the analysis to include more information so that there will be fewer landscape features listed without a complete set of applicable ecosystem service value.

ACKNOWLEDGEMENTS

This work was supported in part by Contract No. SR04-075, "Valuation of New Jersey's Natural Capital" from the New Jersey Department of Environmental Protection, William Mates, Project Officer.

REFERENCE

Azar, C. and T. Sterner (1996). "Discounting and distributional considerations in the context of global warming." Ecological Economics **19**(2): 169-184.

Bergstrom, J. C. and P. De Civita (1999). "Status of benefits transfer in the United States and Canada: A review." Canadian Journal of Agricultural Economics-Revue Canadienne d'Agroeconomie **47**(1): 79-87.

Bishop, R. C., P. A. Champ, et al. (1997). Measuring non-use values: theory and empirical applications. Determining the value of non-marketed goods. R. J. Kopp, W. W. Pommerehne and N. Schwarz, Kluwer Academic Publishers. **59-81**.

Boumans, R., R. Costanza, et al. (2002). "Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model." Ecological Economics **41**(3): 529-560.

Brookshire, D. S. and H. R. Neill (1992). "Benefit Transfers - Conceptual and Empirical Issues." Water Resources Research **28**(3): 651-655.

Brouwer, R. (2000). "Environmental value transfer: state of the art and future prospects." Ecological Economics **32**(1): 137-152.

Champ, P. A., K. J. Boyle, et al., Eds. (2003). A primer on nonmarket valuation. Dordrecht, Kluwer Academic Press.

Costanza, R. and H. E. Daly (1992). "Natural Capital and Sustainable Development." Conservation Biology **6**(1): 37-46.

Costanza, R., S. C. Farber, et al. (1989). "Valuation and management of wetland ecosystems." Ecological Economics **1**(4): 335-361.

Costanza, R. and C. Folke (1997). Valuing ecosystem services with efficiency, fairness and sustainability as goals. Nature's services: societal dependence on natural ecosystems. G. Daily. Washington, D.C., Island Press**:** 49-68.

Daily, G. (1997). Nature's services: societal dependence on natural ecosystems. Washington, D.C., Island Press.

de Groot, R. S., M. A. Wilson, et al. (2002). "A typology for the classification, description and valuation of ecosystem functions, goods and services." Ecological Economics **41**(3): 393-408.

Desvousges, W. H., M. C. Naughton, et al. (1992). "Benefit Transfer - Conceptual Problems in Estimating Water- Quality Benefits Using Existing Studies." Water Resources Research **28**(3): 675-683.

Eade, J. D. O. and D. Moran (1996). "Spatial economic valuation: Benefits transfer using geographical information systems." Journal of Environmental Management **48**(2): 97-110.

EPA, U. S. (2000). Guidelines for Preparing Economic Analyses. EPA 240-R-00-003. . Washington, D.C., U.S. Environmental Protection Agency.

Farber, S., R. Costanza, et al. (2006). "Linking ecology and economics for ecosystem management." Bioscience **56**(2): 121-133.

Freeman III, A. K. (2003). The measurement of environmental and resources values. Washington, DC, Resource for the Future.

Griffiths, C. (2002). "The use of benefit-cost analyses in environmental policy making." from http://www.bus.ucf.edu/mdickie/Health%20Workshop/Papers/Griffiths.pdf.

Howarth, R. B. and S. Farber (2002). "Accounting for the value of ecosystem services." Ecological Economics **41**(3): 421-429.

Limburg, K. E., R. V. O'Neill, et al. (2002). "Complex systems and valuation." Ecological Economics **41**(3): 409-420.

Moran, D. (1999). "Benefits transfer and low flow alleviation: what lessons for environmental valuation in the UK?" Journal of Environmental Planning and Management **42**(3): 425-436.

Newell, R. G. and W. A. Pizer (2003). "Discounting the distant future: how much do uncertain rates increase valuations?" Journal of Environmental Economics and Management **46**(1): 52-71.

Nordhaus, W. (2007). "ECONOMICS: Critical Assumptions in the Stern Review on Climate Change." Science **317**(5835): 201-202.

Rosenberger, R. S. and T. D. Stanley (2006). "Measurement, generalization, and publication: Sources of error in benefit transfers and their management." Ecological Economics **60**(2): 372-378.

Stern, N. and C. Taylor (2007). "ECONOMICS: Climate Change: Risk, Ethics, and the Stern Review." Science **317**(5835): 203-204.

Sumaila, U. R. and C. Walters (2005). "Intergenerational discounting: a new intuitive approach." Ecological Economics **52**(2): 135-142.

Tol, R. S. J. (1999). "The marginal costs of greenhouse gas emissions." Energy Journal **20**(1): 61-81.

Troy, A. and M. A. Wilson (2006). "Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer." Ecological Economics **60**(2): 435-449.

Turner, R. K., J. Paavola, et al. (2003). "Valuing nature: lessons learned and future research directions." Ecological Economics **46**(3): 493-510.

Weitzman, M. L. (1998). "Why the far-distant future should be discounted at its lowest possible rate." Journal of Environmental Economics and Management **36**(3): 201-208.

Wilson, M., A. Troy, et al. (2004). The economic geography of ecosystem goods and services: revealing the monetary value of landscapes through transfer methods and Geographic Information Systems. Cultural Landscapes and Land Use. M. Dietrich and V. D. Straaten, Kluwer Academic.

Wilson, M. A. and S. R. Carpenter (1999). "Economic Valuation of Freshwater Ecosystem Services in the United States 1971-1997." Ecological Applications **9**(3): 772-783.

Wilson, M. A. and R. B. Howarth (2002). "Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation." Ecological Economics **41**(3): 431-443.

Table 1: New Jersey Land Cover Typology

Land Cover Type
Beach
Coastal Shelf
Cropland
Estuary and tidal bay
Forest
Freshwater wetland
Open water
Pasture/grassland
Riparian zone
Saltwater wetland
Urban greenspace
Urban or barren
Woody perennial

Table 2: Gap Analysis of Valuation Literature

	Fresh Wetland	Salt Wetland	Estuary	Open Freshwater	Beach	Riparian Buffer	Forest	Cropland	Urban Green	Pasture	Coastal Shelf
Gas & climate regulation							31		3	1	
Disturbance prevention		2			2	2					
Water regulation	1								1		
Water supply	6		3	5		9	1				2
Soil retention & formation										1	
Nutrient regulation											
Waste treatment		3									
Pollination							1	2			
Biological control											
Refugium function & wildlife conservation	1	4	5				8				
Aesthetic & Recreational	7	3	9	14	4	8	14	2	3	2	
Cultural & Spiritual		1			1	1					

Total $ Estimates: 163
Total Studies: 94

Table 3: Summary of Average Value of Annual Ecosystem Services (2004 US$ $acre^{-1}$ yr^{-1})

Notes:

1. Row and column totals are in acre$ yr^{-1} i.e. Column totals ($/yr) are the sum of the products of the per acre services in the table and the area of each land cover type, not the sum of the per acre services themselves.
2. Shaded cells indicate services that do not occur or are known to be negligible. Open cells indicate lack of available information.

Land Cover	Area (acres)	Gas/Climate Regulation	Disturbance Regulation	Water Regulation	Water Supply	Soil Formation	Nutrient Cycling	Waste Treatment	Pollination	Biological Control	Habitat/Refugia	Aesthetic & Recreation	Cultural & Spiritual	Totals
Coastal & Marine	**953,892**													
Coastal Shelf	299,835				620									$620
Beach	7,837		27,276									14,847	24	$42,147
Estuary	455,700				49						364	303		$715
Saltwater Wetland	190,520		1					6,090			230	26	180	$6,527
Terrestrial	**4,590,281**													
Forest	1,465,668	60			9				162		923	130		$1,283
Grass/Rangelands	583,009	5				6						1		$12
Cropland	90,455								8			15		$23
Freshwater Wetlands	814,479			5,957	1,161						5	1,571		$8,695
Open Fresh Water	86,232				409							356		$765
Riparian Buffer	15,146		88		1,921							1,370	4	$3,382
Urban Greenspace	169,550	336		6								2,131		$2,473
Urban or Barren	1,365,742													$0
Total	5,544,173	147,511,220	215,245,657	4,852,967,357	1,231,742,644	3,398,941	0	1,160,212,484	238,418,048	0	1,565,783,385	2,143,849,095	34,559,302	11,446,176,912

Table 4: Total Acreage and Mean Flow of Ecosystem Services in New Jersey

Name	Acreage	ESV Flows using A studies
Coastal and Marine		
Coastal Shelf	299,835	$185,843,730
Beach	7,837	$330,322,259
Estuary and Tidal Bay	455,700	$325,989,335
Saltwater Wetland	190,520	$1,243,545,862
Terrestrial		
Forest	1,465,668	$1,880,935,494
Pasture/grassland	583,009	$6,751,242
Cropland	90,455	$2,103,089
Freshwater Wetland	814,479	$7,081,746,098
Open Fresh Water	86,232	$65,993,537
Riparian Buffer	15,146	$51,230,004
Urban Greenspace	169,550	$419,227,482
Urban or Barren	1,365,742	$0
TOTAL	5,544,173	**$11,593,688,132**

Table 5: Net Present value (NPV) of Annual Flows of Ecosystem Services Using Various Discount Rates and Discounting Techniques

Annual Flow Value (Billion$/yr)	**0%, 100 yrs**	**1%**	**3%**	**5%**	**8%**
	Standard constant discount rate				
$11	$1,100	$1,100	$367	$220	$138
	Declining discount rate (300 yr time frame)				
$11		$1,809	$640	$299	$151
	Intergenerational Discounting				
$11		$5,542	$870	$405	$212

Table 6: The Standard Deviation of Transferred Estimates for Ecosystem Services

Land Cover	Gas/Climate Regulation	Disturbance Regulation	Water Regulation	Water Supply	Soil Formation	Nutrient Cycling	Waste Treatment	Pollination	Biological Control	Habitat/Refugia	Aesthetic & Recreation	Cultural & Spiritual
Coastal Shelf				146 (2, 2)								
Beach		9,139 (2, 2)									18067 (4, 4)	0 (1, 1)
Estuary				41 (3, 3)						548 (2, 5)	448 (4, 9)	
Saltwater Wetland		0 (2, 2)					9098 (1, 3)			274 (4, 4)	25 (3, 3)	0 (1, 1)
Forest	103 (13, 31)			0 (1, 1)				0 (1, 1)		1211 (5, 8)	204 (9, 14)	
Grass/Rangelands	0 (1, 1)				0 (1, 1)						1 (2, 2)	
Cropland								4 (2, 2)			15 (2, 2)	
Freshwater Wetlands			0 (1, 1)	1183 (5, 6)						0 (1, 1)	1600 (5, 8)	
Open Fresh Water				234 (5, 5)							310 (9, 14)	
Riparian Buffer		50 (1, 2)		3704 (8, 9)							2150 (7, 8)	0 (1, 1)
Urban Greenspace	424 (2, 3)		0 (1, 1)								1189 (1, 3)	
Urban or Barren												

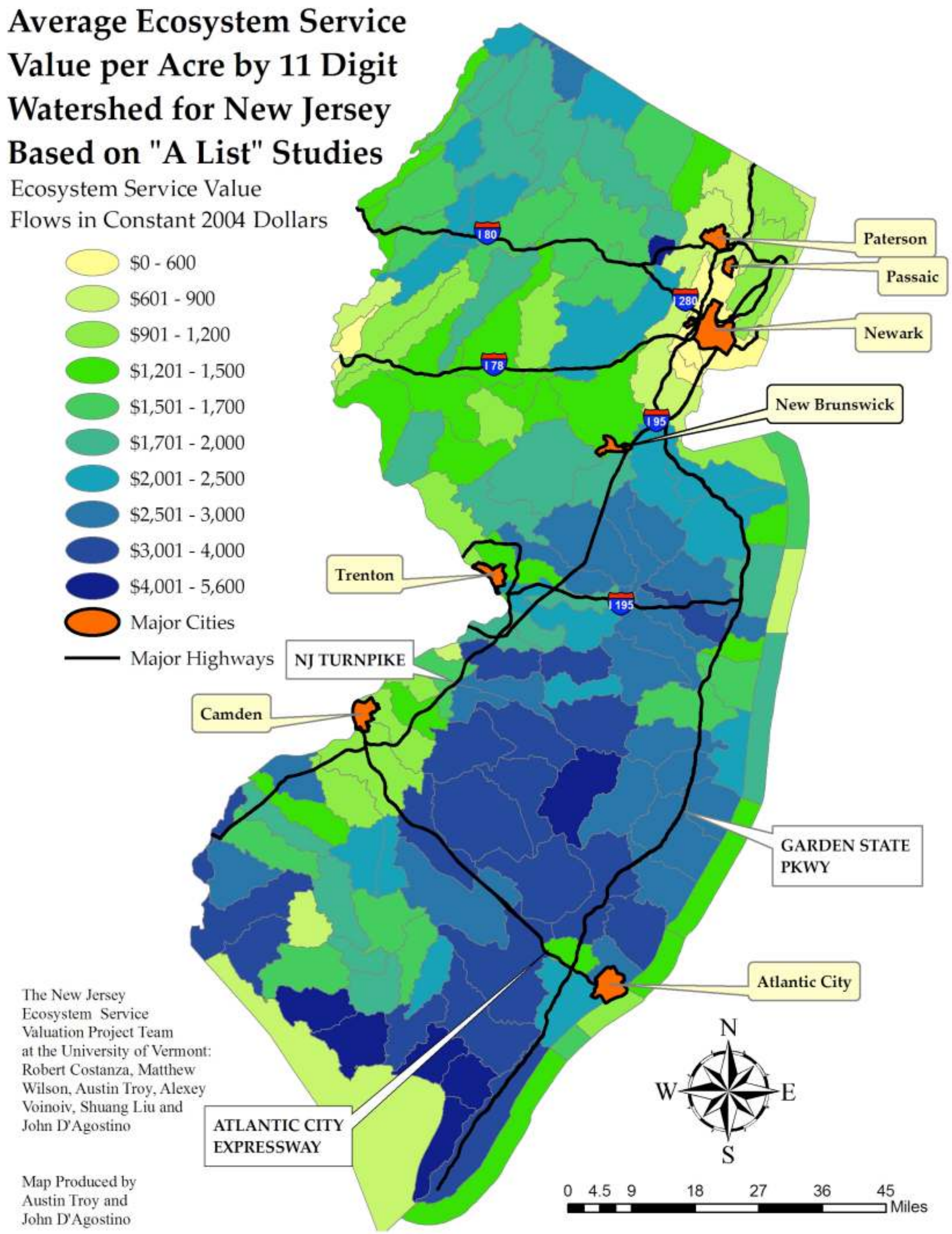

Figure 1: Average Ecosystem Service Value per acre by Watershed for New Jersey

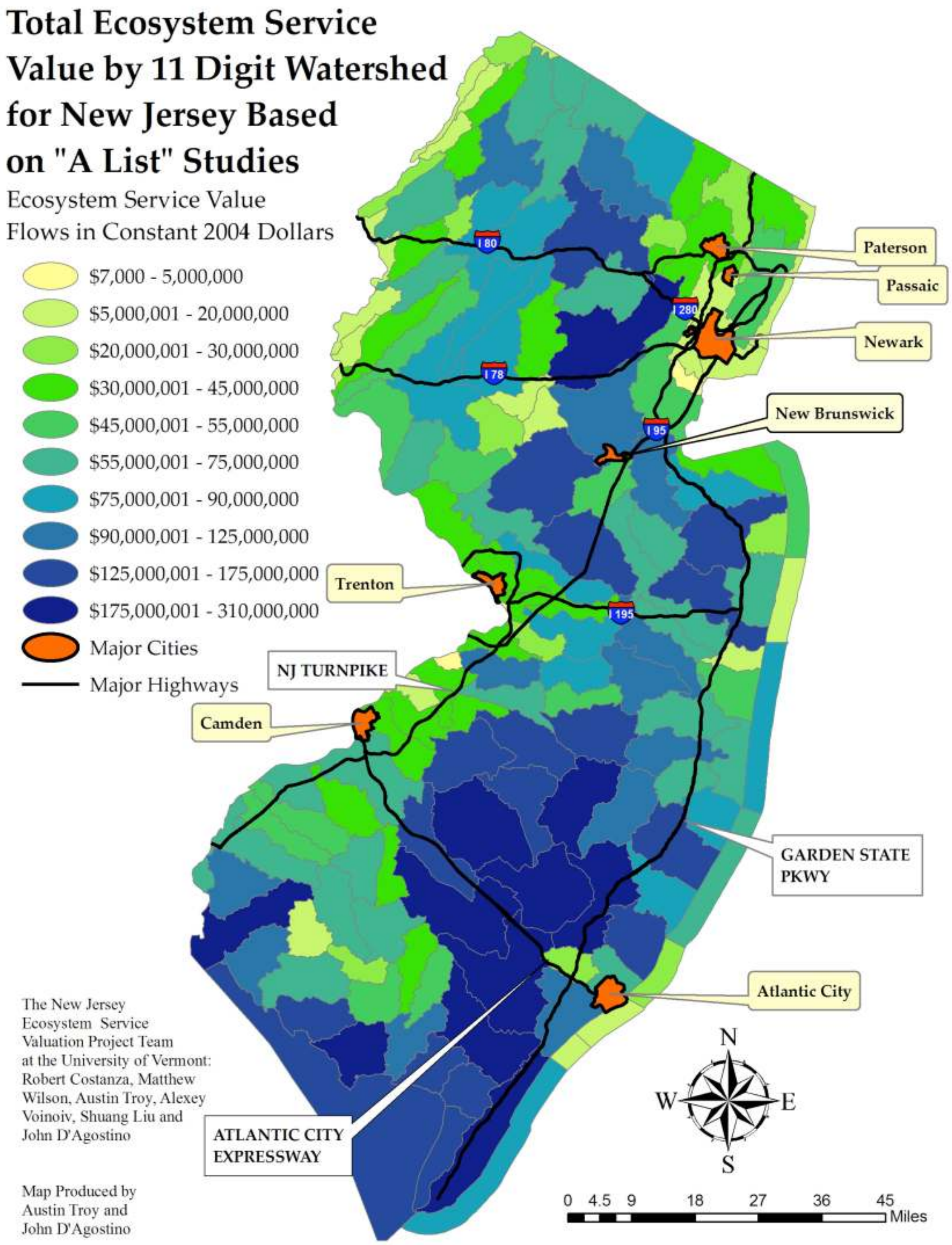

Figure 2: Total Ecosystem Service Value by watershed for New Jersey

Evaluating the Non-Market Value of Ecosystem Goods and Services provided by Coastal and Nearshore Marine Systems*

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Keywords: Ecosystem service, Coastal and marine systems, Non-market valuation

* This paper is in press as a book chapter in Pattterson and Glavovic (eds.) *Ecological Economics of the Oceans and Coasts.*

Abstract

The goods and services provided by coastal and nearshore marine systems and the natural capital stocks that produce them contribute significantly to human welfare, both directly and indirectly, and therefore represent a potentially significant portion of the total economic value of the global environment. Marine and coastal systems including sea-grass beds, coastal wetlands, mangroves and estuaries are particularly rich in ecosystem services. They provide a wide range of highly valued resources including fisheries, wildlife habitat, nutrient cycling, and recreational opportunities. In this chapter, we present a conceptual framework for the assessment and non-market valuation of ecosystem services provided by coastal and marine systems. First, building on recent developments by the UN-Sponsored Millennium Ecosystem Assessment we elucidate a formal system based on functional diversity for classifying and valuing coastal and nearshore marine ecosystem services, emphasizing that no single ecological or economic methodology can capture the total value of these complex systems. Second, we demonstrate the process of ecosystem service valuation using a series of economic case studies and examples drawn from peer-reviewed literature. We conclude with observations on the future of coastal and nearshore marine ecosystem service valuation and its potential role in the science and management of oceanic zone resources.

1. *Introduction*

Throughout history, humans have favored coastal and nearshore marine locations as desirable places to live, work, and play. Forming a dynamic zone of convergence between land and sea, the coastal and marine regions of the earth serve as unique geological, ecological and biological domains of vital importance to a vast array of terrestrial and aquatic life (Argady *et al* 2005; Wilson *et al* 2005). Given this abundance, it is perhaps not surprising that the coastal zone (≤ 150km of the coastline) has long served as a focal point for human activity on planet earth.

Early on, estuaries and inlets served as places of relative shelter that also provided staging areas for harvesting food and fibre. As trading between human settlements developed, ports grew up in those places that offered sea-going vessels protection and provided access to the interior via freshwater river systems. The industrial revolution increased the use of the coastal zone not only for the transport of raw materials and finished goods, but also in new uses such as water extraction and the discharge of waste. With the ascendance of late-industrial society, recreational aspects of the coastal zone have increased in importance, as inland waterways, stretches of beach, coral reefs and rocky cliffs provide opportunities for leisure activity.

Coastal areas around the world are currently undergoing significant human population growth pressures (Argady *et al* 2005). Approximately 44% of the global population in 1994 lived within 150 km of a coastline (Cohen *et al* 1997). Today, that trend appears to be accelerating. Already, more than half of the United States population lives along the coast and in coastal watersheds (Beach 2002). Coastal states in the U.S. are among the nation's fastest growing and are expected to experience most of the absolute growth in population in the decades ahead (Beatley *et al* 2002). The overwhelming majority of Chinese (94%) live in the eastern third of China and over 56% reside in coastal provinces along the Yangtze river valley, and two coastal municipalities—Shanghai and Tianjin (Hinrichsen 1998). In Europe, according to projections worked out by the Mediterranean Blue Plan (http://www.planbleu.org/indexa.htm), the Mediterranean Basin's resident population could go as high as 555 million by 2025. These projections clearly show that coastal regions within the Mediterranean could reach 176 million—30 million more than the entire coastal population in 1990.

Today, there are few, if any, coastal regions that have not been affected in some way by human intervention (Argady *et al* 2005; Vitousek *et al* 1997; Wilson *et al* 2005). Just the fact that so many people live in the coastal zone is a form of pressure on the natural structures and processes that provide the goods and services people desire. Moreover, humans are now a major agent influencing the morphology and ecology of the coastal zone either directly by means of engineering and construction works and/or indirectly by modifying the physical, biological and chemical processes at work within the coastal system (Townend 2002).

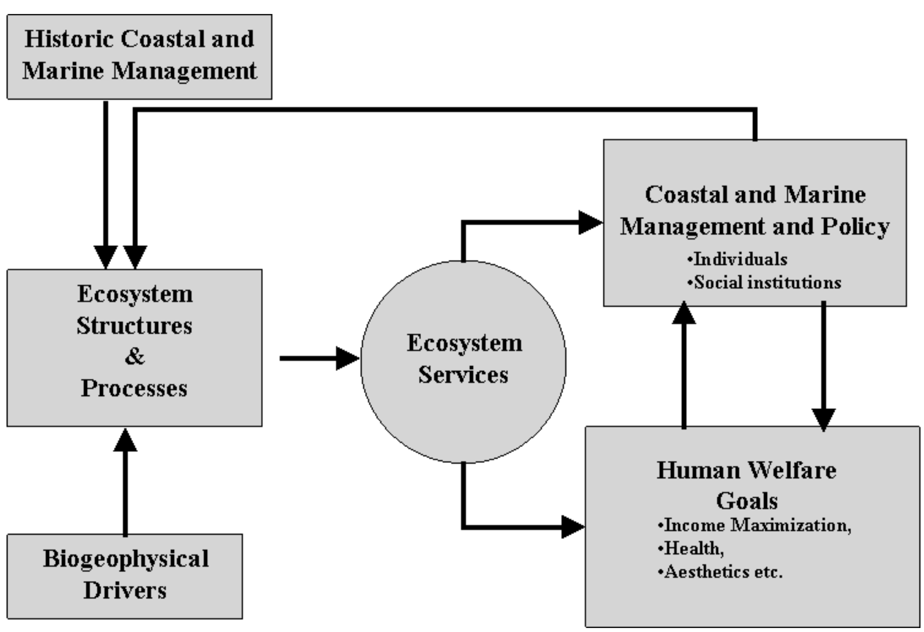
The population and development pressures that coastal and nearshore marine areas are now experiencing raise significant challenges for coastal planners and decision makers. Communities must often choose between competing uses of the coastal environment and the myriad goods and services provided by healthy, functioning ecosystems. Should this shoreline be cleared and stabilized to provide new land for development, or should it be maintained in its current state to serve as wildlife habitat? Should that coastal wetland be drained and converted to agriculture or should more wetland area be created to provide freshwater filtration services? Should this coral reef be mined the production of lime, mortar and cement or should it be sustained to provide renewable seafood products and recreational opportunities?

To choose from among these competing options, it is important to know not only what ecosystem goods and services will be affected but also what they are actually worth to different members of society (Farber *et al* 2006). When confronting decisions that pit different ecosystem services against one another, decision makers cannot escape making a *social choice* based on values: whenever one alternative is chosen over another, that choice indicates which alternative is deemed to be worth more than other alternatives. In short, "we cannot avoid the valuation issue, because as long as we are forced to make choices, we are doing valuation" (Costanza & Folke 1997) p. 50). In this chapter, we show that efforts to assess and quantify *all* the benefits associated with coastal ecosystem goods and services will be necessary for policy and managerial decisions that maximize social interests that benefit from the characteristics of such complex systems.

2. Conceptual Framework

Coastal and nearshore marine systems including fish nurseries, coral reef systems, estuaries, wetlands and sandy beaches provide many different ecosystem goods and services to human society. An ecosystem service, by definition, contains "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Daily 1997). Ecosystem goods, on the other hand, represent the material products that are obtained from natural systems for human use (DeGroot *et al.* 2002). Ecosystem goods and services occur at multiple scales, from climate regulation and carbon sequestration at the global scale, to flood protection, water supply, soil formation, nutrient cycling, waste treatment and pollination at the local and regional scales (DeGroot *et al* 2002; Heal *et al* 2005). They also span a range of degree of direct connection to human welfare, with those listed above being less directly connected, while food, raw materials, genetic resources, recreational opportunities, and aesthetic and cultural values are more directly connected. For this reason, ecologists, social scientists and environmental managers are increasingly interested in assessing the human welfare goals associated with coastal and marine ecosystem goods and services (Argady *et al* 2005; Barbier 2000; Farber *et al* 2006; Wilson *et al* 2005).

Figure 1: Framework for Integrated Assessment and Valuation of Ecosystem Functions, Goods and Services in the Coastal and Marine Zone*



*Adapted from Wilson *et. al.* (2005)

Fig. 1 represents an integrated framework the authors have developed for the assessment of ecosystem goods and services within the coastal and nearshore marine environment, including consideration of ecological structures and processes, land use decisions, human welfare and the feedbacks between them. As the schematic shows, ecosystem goods and services form a pivotal conceptual link between human and ecological systems. Ecosystem structures and processes are influenced by long-term, large-scale biophysical drivers which in turn create the necessary conditions for providing the ecosystem goods and services people value.

The concept of ecosystem goods and services used in this chapter is inherently *anthropocentric*: it is the presence of human beings as welfare-maximizing agents that enables the translation of basic ecological structures and processes into value-laden entities. Through laws and rules, land use management and policy decisions, individuals and social groups make tradeoffs between these values. In turn, these land use decisions directly modify the structures and processes of the coastal zone by engineering and construction and/or indirectly by modifying the physical, biological and chemical processes of the natural system (Boumans *et al* 2002).

In this chapter, we use the concept of ecosystem goods and services to describe a diversity of human values associated with coastal systems (Farber *et al* 2002). We focus on peer-reviewed estimates of non-market economic values and discuss how these values can be used to inform decisions about the future of the coastal and marine environments.

3. Classifying Ecosystem Goods and Services in Coastal and Marine Systems

Coastlines and marine systems around the world exhibit a variety of physical types and characteristics, the result of differences in geophysical and biophysical processes.

There are also a number of distinct habitat and ecosystem types within the coastal and nearshore zone, each suggesting unique management and planning needs. As mentioned previously, coastal and marine regions are dynamic interface zones where land, water and atmosphere interact in a fragile balance that is constantly being altered by natural and human influences. When establishing classification schemes for the coastal and marine zone, it is important to remember that critical biological and physical drivers and interconnections extend beyond these areas and that coastal zones can be significantly affected by events that happen great distances (temporal and spatial) from the coast itself.

Accurate land cover/land use definition and classification are essential preliminary steps in the valuation and management of coastal systems. In this chapter, we adopt a land use classification system with a high level of standardization that builds on previous work by the authors (Wilson *et al* 2005; Wilson *et al* 2004). In Table 1 below, we have identified specific coastal and nearshore features using this typology.

For example, nearshore ocean is distinguished here from open ocean by those ocean areas are either 50m in depth or 100km offshore. Nearshore islands and nearshore open space analogously fall within the 100km zone offshore or inshore from the physical coastline respectively. Estuaries and lagoons are classified as those highly productive areas in the nearshore environment where mixing between salt and freshwater take place. Saltwater wetlands, marshes or salt ponds are distinguished from the former by the fact that they occur inland of the physical coastline. Beaches or dunes may occur on nearshore islands or within nearshore open space, but are given a distinct class of their own due to the significant value attached to them by humans. Analogously, coral reefs or coral atolls are distinguished from nearshore islands (Moberg & Folke 1999). Finally, both mangrove systems and seagrass beds or kelp forests are recognized as a separate land cover class due to their unique ecological features and high levels of productivity (Barbier 2000).

Accurate definition and classification of ecosystem goods and services are also essential preliminary steps in the valuation of coastal and marine systems. In this chapter, we adopt a modified version of the newly standardized system developed in the UN-sponsored Millennium Ecosystem Assessment *(Millennium Ecosystem Assessment 2003)*and adapt that system to a typology of ecosystem goods and services developed in collaboration with colleagues (DeGroot *et al* 2002; Farber *et al* 2006). The Millennium Assessment (2003) provides a useful way of grouping ecosystem goods and services into four basic categories based on their functional characteristics:

1. *Regulating Services*: ecosystems regulate essential ecological processes and life support systems through bio-geochemical cycles and other biospheric processes. These include things like disturbance prevention and flood control.

2. *Cultural Services:* ecosystems provide an essential 'reference function' and contribute to the maintenance of human health and well being by providing spiritual fulfillment, historic integrity, recreation and aesthetics.

3. *Supporting Services:* ecosystems also provide a range of services that are necessary for the production of the other three service categories. These include nutrient cycling, soil formation and habitat functions.

4. *Provisioning Services:* the provisioning function of ecosystems supply a large variety of marketed ecosystem goods and other services for human consumption, ranging from food and raw materials to energy resources.

As this list shows, not all ecosystem goods and services are the same--there is no single category that captures the diversity of what functioning coastal and marine systems' provide humans. In Table 1, we identify studies in the valuation literature, match them with relevant landscape features and ecosystem goods and services to create a matrix of the best available data. Since this chapter is focused on the non-market values, *provisioning* services are left out of this analysis.

Table 1: Non-Market Services in Coastal and Marine Systems

					Supporting services						Regulating services						Cultural services
	Nutrient cycling	Net primary production	Pollination and seed dispersal	Habitat	Hydrological cycle	Gas and Climate regulation	Disturbance Regulation	Biological regulation	Water regulation	Soil retention	Waste regulation	Nutrient regulation	Water supply	Recreation	Aesthetic	Science and education	Spiritual and historic
Estuaries and Lagoons				2									9	6	5		
Beaches and Dunes				1			2						7	11	1		3
Saltwater Wetlands		1		3			2						4	9	3		1
Nearshore Freshwater Wetlands							1		3				1	5	1		
Seagrass or Kelp beds				1				1							1		
Nearshore Islands				2			1							1			
Coral Reefs and Atolls													1	8			
Mangrove				1													
Semi-enclosed Seas				2											1		
Open Ocean																	
Nearshore Ocean				4									5	24	1		
Nearshore Open Space				1									4	13	2		

Total Studies: 70

Observations: 155

The information depicted in Table 1 shows that ecosystem service values can be associated with a variety of landscape features or habitats or both. Numbers in the table represent ecosystem goods and services that have been *empirically measured* in the economic valuation literature and the number of observations associated with each land cover-ecosystem service pairing. The quantitative results for each of the 70 studies (155 observations) are presented in greater detail in the technical appendix that follows this chapter.

4. Valuation of Coastal and Nearshore Marine Ecosystem Services

In economic terms, the ecosystem goods and services depicted above in Table 1 and the technical appendix yield a number of important values to humans. When discussing these values, however, we first need to clarify what the underlying concept actually means (Farber *et al* 2002). The term 'value' as it is employed in this chapter has its conceptual foundation in economic theory (Freeman 1993). In this limited sense, value can be reflected in two theoretically commensurate empirical measures. First, there is the amount of money people are willing to pay for specific improvements in a good or service, *willingness to pay* (WTP). Second, there is the minimum amount an individual would need to be compensated to accept a specific degradation in a good or service, *willingness to accept compensation* (WAC) (Bishop *et al* 1997). Simply put, economic value is the amount of money a person is willing to give up in order to get a thing, or the amount of money required to give up that thing. To date in the literature, WTP has been the dominant measure of economic value. However, WTP is not restricted to what we actually observe from people's transactions in a market. Instead, “it expresses how much people would be willing to pay for a given good or service, whether or not they actually do so” (Goulder & Kennedy 1997).

A central concern in coastal management is one of making social tradeoffs--allocating scarce resources among society’s members. For example, if society wished to make the most of its endowment of coastal resources, it should be possible to compare the value of what society's members receive from any improvement in a given coastal ecosystem with the value of what its members give up to degrade the same system. The prevailing approach to this type of assessment in the literature is cost-benefit analysis (Ableson 1979; Kneese 1984; Turner 2000). Cost-benefit analysis is characterized by a fairly strict decision-making structure: “defining the project, identifying impacts which are economically relevant, physically quantifying impacts as benefits or costs” and then, “calculating a summary monetary valuation” (Hanley & Spash 1993). Given this approach, a key question comes down to: what gets counted?

In addition to the production of marketable goods, coastal ecosystems provide natural functions such as nutrient recycling as well as conferring aesthetic benefits to humans (Costanza *et al* 1997). Coastal goods and services may therefore be divided into two general categories: (1) the provision of direct *market* goods or services such as food, transportation, electricity generation, and pollution disposal; and, (2) the provision of *non-market* goods or services which include things like biodiversity, support for terrestrial and estuarine ecosystems, habitat for plant and animal life, and the satisfaction people derive from simply knowing that a beach or coral reef exists.

The market value of ecosystem goods and services are the observed trading ratios for services that are directly traded in the marketplace: price = exchange value. The exchange-based, welfare value of a natural good or service is its market price net of the cost of bringing that service to market. For example, the exchange-based value of fresh fish to society is based on its catch rate and "value at landing" which is the market price of fish, minus harvest and time management costs. Estimating exchange-based values in this case is relatively simple, as observable trades exist from which to measure value.

Since individuals can be observed making choices between objects in the marketplace while operating within the limits of income and time, economists have developed several market-based measures of value as imputations from these observed choices. While monetary measures of value are not the only possible yardstick, they are convenient since many choices involve the use of money. Hence, if you are observed to pay $9 for a pound of shrimp, the imputation is that you value a pound of shrimp to be at least $9, and are willing to make a trade-off of $9 worth of other things to obtain that shrimp. The money itself has no intrinsic value, but represents other things you could have purchased. Time is often considered another yardstick of value; if someone spends 2 hours fishing, the imputation is that the persons values the fishing experience to be worth more than 2 hours spent on other activities. Value is thus a resultant of the expressed tastes and preferences of persons, and the limited means with which objects can be pursued. As a result, the scarcer the object is, the greater its value will be on the margin.

By estimating the economic value of ecosystem goods and services not traded in the marketplace, however, social costs or benefits that otherwise would remain hidden or unappreciated are revealed. While measuring exchange values requires monitoring market data for observable trades, non-market values of goods and services are much broader and more difficult to measure. Indeed, it is these values that are have captured the attention of environmental and resource economists who have developed a number of techniques for valuing ecosystem goods and services (Bingham *et al* 1995; Freeman 1993). When there are no explicit markets for services, more indirect means of assessing economic values must be used. A spectrum of economic valuation techniques commonly used to establish the WTP or WTA when market values do not exist are identified below.

•**Avoided Cost** (AC): services allow society to avoid costs that would have been incurred in the absence of those services; flood control provided by barrier islands avoids property damages along the coast.

•**Replacement Cost** (Pearce): services could be replaced with man-made systems; nutrient cycling waste treatment can be replaced with costly treatment systems.

•**Factor Income** (FI): services provide for the enhancement of incomes; water quality improvements increase commercial fisheries catch and incomes of fishermen.

•**Travel Cost** (TC): service demand may require travel, whose costs can reflect the implied value of the service; recreation areas attract distant visitors whose value placed on that area must be at least what they were willing to pay to travel to it.

• **Hedonic Pricing** (HP): service demand may be reflected in the prices people will pay for associated goods: For example, housing prices along the coastline tend to exceed the prices of inland homes.

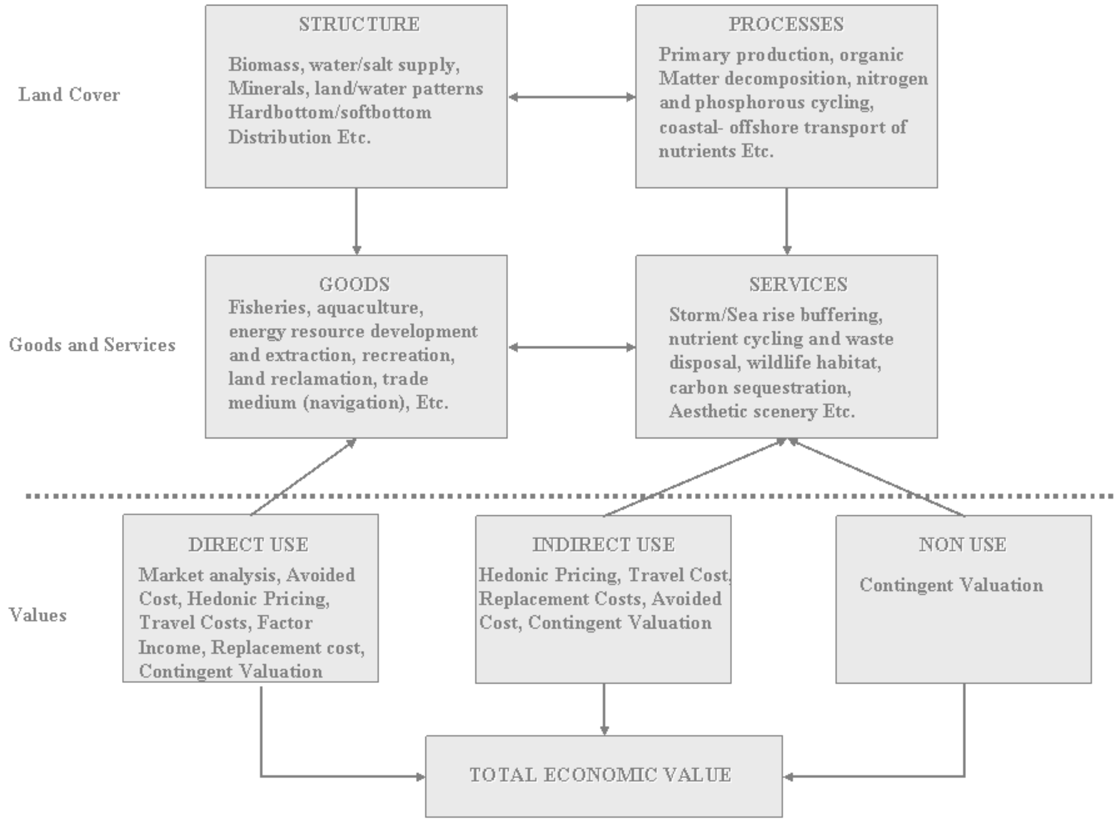
•**Marginal Product Estimation** (MP): Service demand is generated in a dynamic modeling environment using production function (i.e., Cobb-Douglas) to estimate value of output in response to corresponding material input.

•**Contingent Valuation** (CV): service demand may be elicited by posing hypothetical scenarios that involve some valuation of alternatives; people would be willing to pay for increased preservation of beaches and shoreline.

•**Group Valuation** (GV): This approach is based on principles of deliberative democracy and the assumption that public decision making should result, not from the aggregation of separately measured individual preferences, but from *open public debate.*

As these brief descriptions suggest, each economic valuation methodology has its own strengths and limitations, thereby restricting its use to a select range of goods and services associated with coastal systems. For example, to Travel Cost (Argady *et al*) is useful for estimating recreation values, and Hedonic Pricing (HP) for estimating coastal property values, but they are not easily exchanged. Rather, a full suite of valuation techniques is required to quantify the economic value of goods and services provided by a naturally functioning coastal ecosystem. By using a range of methods for the same site, the so-called “total economic value” of a given coastal ecosystem can thus be estimated (Freeman 1993).

Fig. 2: Total Economic Value of Coastal Zone Functions, Goods and Services*



* Adapted from Turner (2000) and Wilson et. al. (2005)

Fig. 2 depicts a model based on the idea of functional diversity, linking different ecosystem structures and processes with the output of specific goods and services, which can then be assigned monetary values using the range of valuation techniques described above. Here, key linkages are made between the diverse structures and processes associated with the coastal zone, the landscape and habitat features that created them, and the goods and services that result. Once delineated, economic values for these goods and services can then be rationally assessed by measuring the diverse set of human preferences for them. In economic terms, the natural assets of the coastal zone can thus yield direct (fishing) and indirect (nutrient cycling) use values as well as non-use (preservation) values of the coastal system. Once accounted for, these values can then be aggregated to estimate the total value of the entire system (Anderson & Bishop 1986).

In principle, a global picture of the potential economic value associated with the coastal zone can be built up via the aggregation of a number of existing valuation studies. For example, in a preliminary estimate of the total economic value of ecosystem services provided by global systems, Costanza *et al.* (1997) showed that while the coastal zone covers only 8% of the world's surface, the goods and services provided by it are responsible for approximately 43% of the estimated total value of global ecosystem services: US$ 12.6 trillion (1997 dollars). While controversial (Pearce 1998; Pimm 1997), this preliminary study made it abundantly clear that coastal ecosystem services do provide an important portion of the total contribution to human welfare on this planet. Furthermore, it demonstrated the need for additional research and indicated the fact that coastal areas are among the most in need of additional study (Costanza 2000).

Such 'environmental benefit transfer' studies often form the bedrock of practical policy analysis because only rarely can policy analysts or managers afford the luxury of designing and implementing an original study for every given ecosystem (Wilson & Hoehn 2006). Instead, decision makers must often rely on the limited information that can be gleaned from past empirical studies that are often quite limited or even contradictory (Desvouges *et al* 1998; Smith 1992). Primary valuation research, while being a 'first best' strategy, is also very expensive and time consuming. Thus, secondary analysis of the valuation literature is a 'second best' strategy that can nevertheless yield very important information in many scientific and management contexts (Rosenberger & Loomis 2000). When analyzed carefully, information from past studies published in the literature can form a meaningful basis for coastal zone policy and management (Beatley *et al* 2002; French 1997). In the final section of this chapter, we demonstrate this integrative approach for coastal and nearshore marine ecosystem service valuation by providing a brief review of case studies drawn from the literature.

5. *Literature Review Results*

Empirical valuation data for coastal ecosystems often appears scattered throughout the scientific literature and is uneven in quality. To elucidate this unevenness, here we present a review of existing non-market valuation literature in order to provide useful insights for further research in the area. All the data discussed here are presented in greater detail in the technical Appendix following this chapter.

Such an exercise provides scientists, coastal managers and business leaders alike with a sense of where the science of coastal ecosystem valuation has come from, and where it might go in the future. Below we synthesize peer-reviewed economic data on coastal ecosystems depicted above in Table 1 and in the technical appendix that follows this chapter. We also have selected a few key examples from the literature for extended discussion. In so doing, we hope to elucidate major findings and gaps in the literature for the reader.

All information presented below and in the technical appendix were obtained from studies that were published in peer-reviewed journal or book chapters. They deal *explicitly* with non-market coastal ecosystem services measured throughout the world. The literature search involved an intensive review of databases on the World Wide Web available at the University of Vermont. Several keywords-- economic value, economics, valuation, management, coastal, marine, wetland, estuary, mangrove, contingent valuation and ecosystem service etc. were combined in various patterns to elicit studies that might be relevant to coastal and marine ecosystem valuation. This search yielded more than 300 citations. Each citation was then located and reviewed by the authors. About 230 citations (>77%) were rejected because they were not peer-reviewed or did not explicitly address the economic valuation of coastal ecosystem goods and services. The literature review yielded a total of 70 and 155 data observations studies for further analysis and discussion.

Results from these studies were sorted by land cover type, ecosystem good and service, valuation methodology and region of study. On this basis, each study was classified as measuring an ecosystem good, service or any combination thereof (several studies report data for more than one good or service). Selected valuation studies for coastal goods or services are discussed in greater detail below.

Figure 3: Valuation Data Distributed by Ecosystem Service

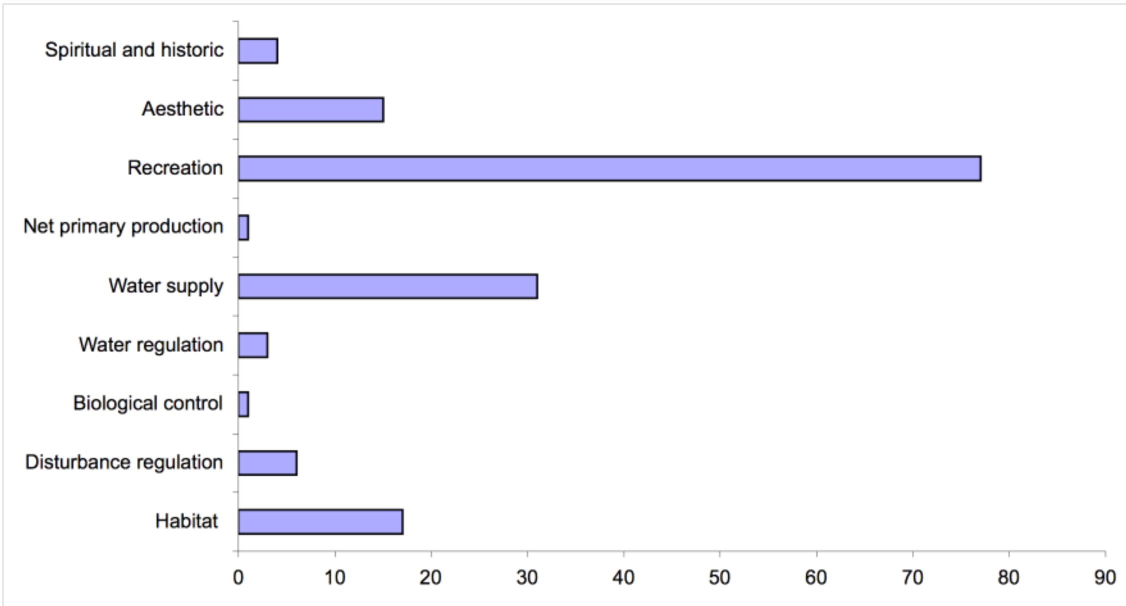


Figure 4: Valuation Data Distributed by Cover Type

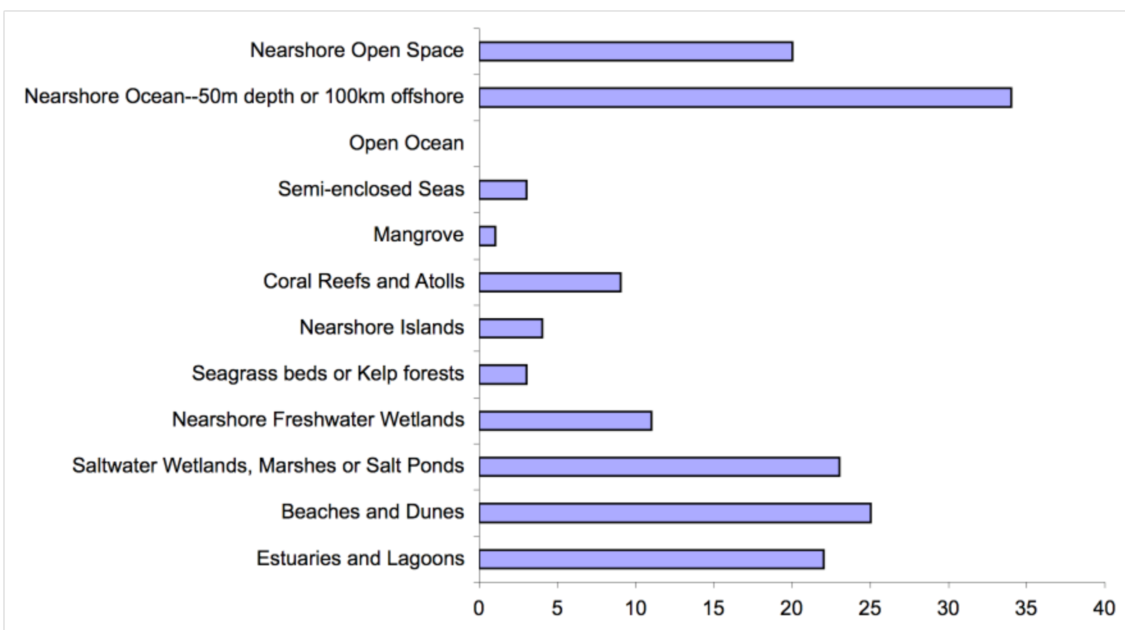
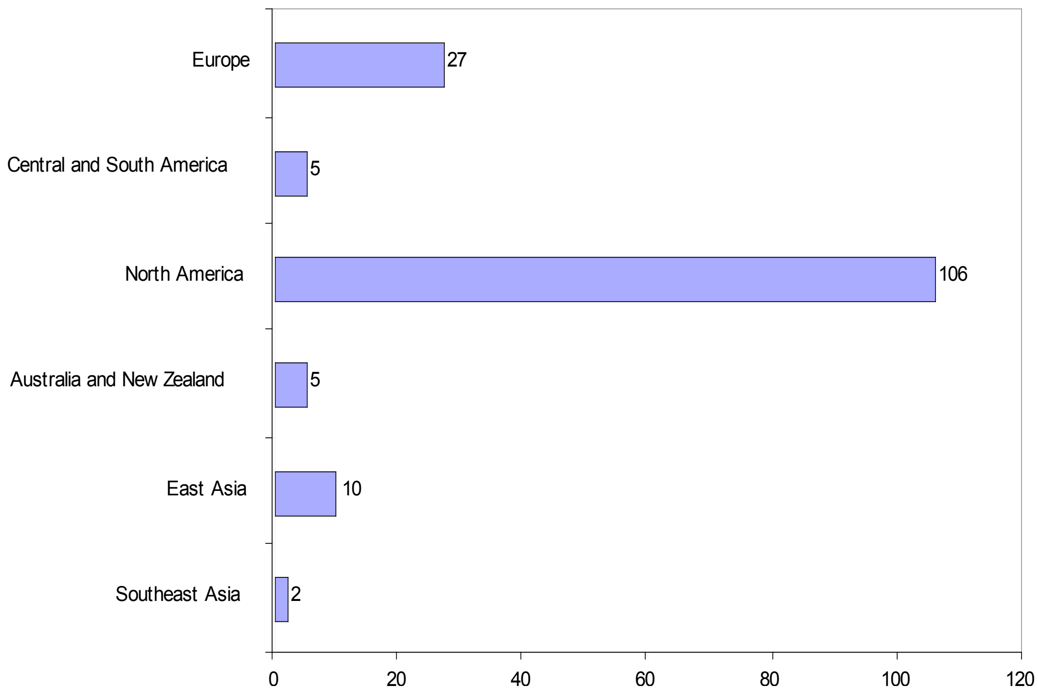


Figure 5: Valuation Data Distributed by Region



Our review of the peer-reviewed literature reveals that many different landscape features and ecological processes within the coastal and nearshore marine zone provide essential natural services to humans, but that the reporting of their economic values remains unevenly distributed. For example, as the pattern of data in Figure 3 confirms, opportunities for recreation and natural amenities (e.g., nearshore fisheries, white sandy beaches) get an inordinate attention in the economic literature while other services such as spiritual and historic or biological control do not get much attention at all. Similarly, as Figure 4 shows, nearshore ocean, open space, freshwater wetlands and saltwater wetlands, marshes or salt ponds have tended to receive the most attention in the peer-reviewed literature while areas such as mangroves and coral reefs have received far more limited attention by economists. Finally, as Figure 5 clearly shows, the vast majority of economic valuation studies in the peer-review literature have been conducted in the

United States with other regions such as Europe, Australia and New Zealand lagging behind. While perhaps not surprising given the early development of environmental and ecological economics as a field of study, the uneven distribution of empirical analyses raises critical issues for decision makers that will need to be addressed in the not-too-distant future (Wilson & Hoehn 2006).

To provide a more in-depth account of the specific types of the non-market valuation literature available today, below we briefly review a select group of published valuation studies reported in table 1 and in the technical appendix and group them according to the type of ecosystem services discussed. As this chapter does not focus specifically on market goods, provisioning services (e.g., food, fuel and fibre) are left out of the analysis. The results from each empirical study are reported in their original monetary metric.

5.1 Supporting Services

As mentioned previously in this chapter, the coastal and nearshore marine environment is one of the most productive habitats in the world. Mangroves, eelgrass, salt marsh and intertidal mud flats all provide a variety of services to the public associated with their nursery and habitat functions. Improvements in the ecological integrity of these habitats may ultimately lead to measurable increases in the production of market goods such as fish, birds and wood products. In other cases, ecological productivity itself can represent a unique class of values not captured by traditional market-based valuation methods. Instead, these values represent an increase in the production of higher trophic levels brought about by the increased availability of habitat (Gosselink *et al* 1974; Turner *et al* 1996). Here, it is critical to realize that one may not, in general, add productivity value estimates to use values estimated using other market-related methodologies (i.e., hedonic and travel cost) because to do so would risk double counting some aspects of value, or measuring the same benefits twice (Desvouges *et al* 1998; Desvousges *et al* 1992).

Valuation studies of the supporting functions provided by coastal and nearshore marine habitat have predominantly focused on the economic value of fishery related services (Barbier 2000; Kaoru *et al* 1995; Lynne *et al* 1981). Most often, the market price of seafood products is used as a proxy when calculating the non-market value of ecosystem goods provided by coastal and nearshore systems.

For example, Farber and Costanza (1987) estimated the productivity of coastal habitat in Terrebonne Parrish, Louisiana, USA by attributing commercial values for several species to the net biomass, habitat, and waste treatment of the wetland ecosystem (Farber & Costanza 1987). Arguing that the annual harvest from an ecosystem is a function of the level of environmental quality, the authors chose to focus on the commercial harvest data for five different native species—shrimp, blue crab, oyster, menhaden, and muskrat—to estimate the marginal productivity of wetlands. The annual economic value (marginal product) of each species was estimated in 1983 dollars: shrimp \$10.86/acre; blue crab \$.67/acre; oyster \$8.04/acre; menhaden \$5.80/acre; and muskrat pelts \$12.09/acre. Taken together, the total value marginal productivity of wetlands in Terrebonne Parrish, Louisiana was estimated at \$37.46 per acre.

In an earlier study, Lynne *et al.* (1981) suggested that the value of the coastal marsh in southern Florida could be modeled by assuming that seafood harvest is a direct function of salt-marsh area. The authors then derived the economic value of a specified change in marsh area through the marginal productivity of fishery harvest. For the blue crab fishery in western Florida salt marshes, a marginal productivity of 2.3 1b per year for each acre of marshland was obtained. By linking the market price of harvested blue crab to this estimate, the authors were able to estimate of the total present value of a marsh acre in human food (blue crab) production at $3.00 for each acre (with a 10% capitalization rate).

5.3 Regulating Services

A critically important service provided by coastal landscapes such as barrier islands, inland wetlands areas, beaches and tidal plains is disturbance prevention. Significant property damages have been attributed to flooding from tidal surges and rainfall as well as wind damage associated with major storm events. For example, Farber (1987) has described an "Avoided Cost" method for measuring the hurricane protection value of wetlands against wind damage to property in coastal Louisiana, USA. Using historical probabilities for storms and wind damage estimates in Louisiana, an expected wind damage function was derived and from this, the author estimated reductions in wind damage from the loss of 1 mile of wetlands. Based on 1983 US dollars, the expected incremental annual damaged from a loss of 1 mile of wetlands along the Louisiana coastline was $69,857 which, when extrapolated to a per-acre estimate, amounts to $.44 per acre (Farber 1987).

In another study, Lindsay et. al. (1992) measured coastal beach visitors' willingness to pay for a beach erosion program in Maine and New Hampshire. Beach erosion has been a substantial problem for many coastal communities, forcing them to choose between active management techniques and the possible loss of valuable waterfront acreage. Willingness to pay was assessed by the authors by asking 985 beach users their willingness to pay for a dedicated fund for a beach protection program in Maine and New Hampshire (Lindsay *et al* 1992). Using a tobit regression technique, the authors estimate average WTP for beach protection of $30.80 per person in 1992 dollars.

5.4 Cultural Services

Stretches of beach, rocky cliffs, estuarine and coastal marine waterways, and coral reefs provide numerous recreational and scenic opportunities for humans. Boating, fishing, swimming, walking, beachcombing, scuba diving, and sunbathing are among the numerous leisure activities that people enjoy worldwide and thus represent significant economic value. Both travel cost (TC) and Contingent Valuation (CV) methods are commonly used to estimate this value. For example, the Chesapeake Bay estuary on the eastern seaboard of the United States has been the focus of an impressive amount of research on nonmarket recreational values associated with coastal systems. When attempting to estimate the monetary worth of water quality improvements in Chesapeake bay, *Bockstael et al.* (1989; p. 2) focused on recreational benefits because it was assumed that most of the increase in well-being associated with such improvements would accrue to recreationists. The authors estimated the average increases in economic value for beach use, boating, swimming, and fishing with a 20% reduction in total nitrogen and phosphorus introduced into the estuary. Using a combination of CV and TC methods, the annual aggregate willingness to pay for a moderate improvement in the Chesapeake Bay's water quality was estimated to be in the range of $10 to $100 million in 1984 dollars (Bocksteal *et al* 1989). In a similar study, Kawabe and Oka (1996) used TC to estimate the aggregate recreational benefit (viewing the bay, clam digging, bathing, sailing, bathing, snorkeling and surfing) from improving notrogen contamination of Tokyo Bay at 53.2 billion yen. Using the CV method, the authors also estimated the aggregate value of improving chemical oxygen demand to reduce the reddish-brown color of the bay at 458.3 billion yen (Kawabe & Oka 1996).

Open space, proximity to clean water, and scenic vistas are often cited as a primary attractor of residents who own property and live within the coastal fringe (Beach 2002). Hedonic pricing (HP) techniques have thus been used to show that the price of coastal housing units vary with respect to characteristics such as ambient environmental quality (i.e., proximity to shoreline, water quality) because buyers will bid up the price of units with more of a desirable attribute (Johnston *et al* 2001). For example, Leggett and Bockstael (2000) use hedonic techniques to show that water quality has a significant effect on property values along the Chesapeake Bay, USA. The authors use a measure of water quality—fecal coliform bacteria counts—that has serious human health implications and for which detailed, spatially explicit information from monitoring is available. The data used in this hedonic analysis consists of sales of waterfront property on the western shore of the Chesapeake Bay that occurred between 1993 and 1997 (Leggett & Bockstael 2000). The authors consider the effect of a hypothetical localized improvement in observed fecal coliform counts--100 counts per 100ml—on a set of 41 residential parcels. The projected increase in property values due to the hypothetical reduction total approximately $230,000. Extending the analysis to calculate an upper bound benefit for 494 properties, the authors estimate the benefits of improving water quality at all sites at $12.145 million (Leggett and Bockstael 2000, p.142).

6. Discussion

Ecosystem goods and services form a fundamental connective link between people and ecological systems. In this chapter, we have shown how coastal and nearshore marine ecosystem goods and services not commonly traded in the marketplace contribute significantly to human welfare. Using an integrated framework developed for the assessment of ecosystem goods and services, we have considered how ecological structures and processes, land use decisions, and human values interact in the coastal and nearshore marine environment. The concept of ecosystem goods and services has thus allowed us to analyze how human beings as welfare-maximizing agents actively translate complex ecological structures and processes into value-laden entities.

The literature reviewed here demonstrates both the opportunities and the challenges inherent in estimating the total economic value of coastal ecosystem goods and services. As the pattern of data in Table 1 suggests, one of the major insights from our analysis is the discrepancy between the ecosystem goods and services that have been documented in the published valuation literature and those that could potentially contribute significantly to human welfare, both directly and indirectly. Accounting for these missing economic values represents a significant challenge for scientists, planners and decision makers involved in coastal zone and nearshore marine management.

The studies presented here and in the technical appendix that follows this chapter further suggest that methodological guidelines and standards are still evolving. Nevertheless, it is evident that within specific contexts, defensible dollar estimates can be obtained and thereby add to the information base for coastal management and decision making. Economic estimates may require considerable creative research and have substantial uncertainties. Yet, the best available data do suggest that indeed, humans attach substantial positive values to the many marketed and non-marketed goods and services coastal systems provide.

Through laws and rules, land use management and policy decisions, individuals and social groups ultimately will make tradeoffs between these values as they continue to live, work and play in the coastal zone. In turn, these land use decisions will directly modify the structures and processes of the coastal zone through engineering and construction and/or indirectly by modifying the physical, biological and chemical processes of the natural system. Resource managers and ecologists should therefore be aware that non-use values have been shown to comprise a sizeable portion of total economic value associated with coastal ecosystems.

As we have shown, assigning economic values to landscape features and habitat functions of coastal and marine ecosystems requires a fuller understanding of the nature of the natural systems upon which they rest. Ecosystem structures and processes are influenced by long-term, large-scale biophysical drivers (i.e., tectonic pressures, global weather patterns) which in turn create the necessary conditions for providing the ecosystem goods and services people value. Ecological information must be thoroughly integrated before a meaningful assessment of economic value can be made. This is a formidable challenge, but we believe that the classification system presented in section 3 of this chapter provides a critical first step.

We conclude with the observation that the most non-market valuation studies to date have been performed for a relatively small subset of coastal and nearshore marine ecosystem goods and services at a limited number of sites in the world. Hence, our ability to generalize from studies presented in this review remains limited but promises to grow as more environmental valuation studies are done (Wilson & Hoehn 2006). The observations and results presented here do provide valuable insights into the challenges and limitations of ecosystem service valuation as it is currently being practiced. The experiences summarized here should be useful to ecologists, managers, and social scientists as they collaborate to estimate the future direction for development in the coastal and nearshore marine environment.

References

Ableson, P. 1979. Cost Benefit Analysis and Environmental Problems. Saxon House. London.

Anderson, G. D., Bishop, R. C. 1986. The Valuation Problem. In *Natural resource economics: policy problems and contemporary analysis*, ed. (D. W. Bromley),89-137 pp. Liuwer Nijoff Publishing. Boston, MA.

Argady, T., Alder, J., Dayton, P., Curran, S., Kitchingman, A., Wilson, M. A., Catenazzi, A., Restrepo, J., Birkeland, C., Blaber, S., Saifullah, S., Branch, G., Boersma, D., Nixon, S., Dugan, P., Davidson, N., Vorosmarty, C. 2005. Coastal Systems: An Assessment of Ecosystem Services. In *Ecosystems and Human Well-Being: Current States and Trends*, ed. (Millennium Ecosystem Assessment),515-49 pp. Island Press. Washington, D.C. .

Barbier, E. B. 2000. Valuing the environment as input: review of applications to mangrove-fishery linkages. Ecological Economics 35: 47-61.

Beach, D. 2002. Coastal Sprawl: The Effects of Urban Design on Aquatic Ecosystems in the United States, Pew Oceans Commission. Arlington, VA.

Beatley, T., Brower, D. J., Schwab, A. K. 2002. An Introduction to Coastal Zone Management. Island Press. Washington, DC.

Bingham, G., Bishop, R., Brody, M., Bromley, D., Clark, E., Cooper, W., Costanza, R., Hale, T., Hayden, g., Kellert, S., Norgaard, R., Norton, B., Payne, J., Russell, C., Suter, G. 1995. Issues in Ecosystem Valuation: Improving Information for Decision Making. Ecological Economics 14: 73-90.

Bishop, R. C., Champ, P. A., Brown, T. C., McCollum, D. W. 1997. Measuring Non-Use Values: Theory and Empirical Applications. In *Determining the Value of Non-Marketed Goods*, ed. (Raymond J. Kopp, Werner W. Pommerehne, Norbert Schwarz),59-81 pp. Kluwer Academic Publishers. Boston.

Bocksteal, N. E., McConnell, K. E., Strand, I. E. 1989. Measuring the Benefits of Improvements in Water Quality: The Chesapeake Bay. Marine Resource Economics 6: 1-18.

Boumans, R. M. J., Costanza, R., Farley, J., Wilson, M. A., Rotmans, J., Villa, F., Portela, R., Grasso, M. 2002. Modeling the Dynamics of the Integrated Earth System and the Value of Global Ecosystem Services using the GUMBO model. Ecological Economics 41: 529-60.

Cohen, J. F., Small, C., Mellinger, A., Gallup, J., Sachs, J. 1997. Estimates of Coastal Populations. Science 278: 1211-2.

Costanza, R. 2000. The Ecological, Economic and Social Importance of the Oceans. In *Seas at the Millenium: An Environmental Evaluation*, ed. (C. R. C. Sheppard),393-403 pp. Pergamon. New York.

Costanza, R., d'Arge, R., deGroot, R., Farber, S., Grasso, M., Hannon, b., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., van den Belt, M. 1997. The Value of the World's Ecosystem Services and Natural Capital. Nature 387: 253-60.

Costanza, R., Folke, C. 1997. Valuing Ecosystem Services with Efficiency, Fairness, and Sustainability as Goals. In *Nature's Services*, ed. (Grechen C. Daily),49-68 pp. Island Press. Washington, DC.

Daily, G. C. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press. Washington D.C.

DeGroot, R., Wilson, M. A., Boumans, R. M. J. 2002. A typology for the classification, description, and valuation of ecosystem functions, goods and services. Ecological Economics 41: 393-408.

Desvouges, W. H., Johnson, F. R., Banzhaf, H. S. 1998. Environmental Policy Analysis with Limited Information: Principles and Applications of the Transfer Method. Edward Elgar. Northhampton MA.

Desvousges, W. H., Naughton, M. C., Parsons, G. R. 1992. Benefit Transfer - Conceptual Problems in Estimating Water- Quality Benefits Using Existing Studies. Water Resources Research 28: 675-83.

Farber, S. 1987. The value of coastal wetlands for the protection of property against hurricane wnd damage. Journal of Environmental Economics and Management 14: 143-51.

Farber, S., Costanza, R. 1987. The Economic Value of Wetlands Systems. Journal of Environmental Management 24: 41-51.

Farber, S., Costanza, R., Childers, D., Erickson, J., Gross, K., Grove, M., Hopkinson, C., Kahn, J. R., Pincetl, S., Troy, A., Warren, P., Wilson, M. A. 2006. Linking Ecology and Economics for Ecosystem Management. Bioscience 56: 117-29.

Farber, S., Costanza, R., Wilson, M. A. 2002. Economic and Ecological Concepts for Valuing Ecosystem Services. Ecological Economics 41: 375-92.

Freeman, M. 1993. The Measurement of Environmental and Resource Values. Resources for the Future. Washington D.C.

French, P. W. 1997. Coastal and Estuarine Management. Routledge. New York, NY.

Gosselink, J. G., Odum, E. G., Pope, R. M. 1974. The value of the tidal marsh, Center for Wetland Resources, Louisiana State University.

Goulder, L. H., Kennedy, D. 1997. Valuing Ecosystem Services: Philosophical Bases and Empirical Methods. In *Nature's Services: Societal Dependence on Natural Ecosystems*, ed. (Gretchen C. Daily),23-48 pp. Island Press. Washington D.C.

Hanley, N., Spash, C. 1993. Cost-Benefit Analysis and the Environment. Edward Elgar Publishing. Cornwall.

Heal, G. M., Barbier, E. B., Boyle, K. J., Covich, A. P., Gloss, S. P., Hershner, C. H., Hoehn, J. P., Pringle, C. M., Polasky, S., Segerson, K., Schrader-Frechette, K. 2005. Valuing Ecosystem Services: Toward Better Environmental Decision-Making. The National Academies Press. Washington D.C. .

Hinrichsen, D. 1998. Coastal Waters of The World: Trends, Threats, and Strategies. Island Press. Washington, DC.

Johnston, R. J., Opaluch, J. J., Grigalunas, T. A., Mazzotta, M. J. 2001. Estimating Amenity Benefits of Coastal Farmland. Growth and Change 32: 305-25.

Kaoru, Y., Smith, V. K., Liu, J. L. 1995. Using Random Utility-Models to Estimate the Recreational Value of Estuarine Resources. American Journal of Agricultural Economics 77: 141-51.

Kawabe, M., Oka, T. 1996. Benefit from improvement of organic contamination of Tokyo Bay. Marine Pollution Bulletin 32: 788-93.

Kneese, A. 1984. Measuring the Benefits of Clean Air and Water. Resources for the Future. Washington D.C.

Leggett, C. G., Bockstael, N. E. 2000. Evidence on the Effects of Water Quality on Residential Land Prices. Journal of Environmental Economics and Management 39: 121-44.

Lindsay, B. E., Halstead, J. M., Tupper, H. C., Vaske, J. J. 1992. Factors Influencing the Willingness to Pay for Coastal Beach Protection. Coastal Management 20: 291-302.

Lynne, G. D., Conroy, P., Prochaska, F. J. 1981. Economic Valuation of Marsh Areas for Marine Production Processes. Journal of Environmental Economics and Management 8: 175-86.

Millennium Ecosystem Assessment. 2003. Ecosystems and Human Well-Being: A Framework for Assessment. Island Press. Washington DC.

Moberg, F., Folke, C. 1999. Ecological Goods and Services of Coral Reef Ecosystems. Ecological Economics 29: 215-33.

Pearce, D. 1998. Auditing the Earth. Environment 40: 23-8.

Pimm, S. L. 1997. The Value of Everything. Nature 387: 231-2.

Rosenberger, R. S., Loomis, J. B. 2000. Using meta-analysis for benefit transfer: In-sample convergent validity tests of an outdoor recreaton database. Water Resources Research 36: 1097-107.

Smith, V. K. 1992. On separating defensible benefit transfers from 'smoke and mirrors'. Water Resources Research 28: 685-94.

Townend, I. 2002. Marine Science for strategic planning and management: the requirement for estuaries. Marine Policy (In Press)

Turner, R. K. 2000. Integrating natural and socio-economic science in coastal management. Journal of Marine Systems 25: 447-60.

Turner, R. K., Subak, S., Adger, W. N. 1996. Pressures, Trends and Impacts in Coastal Zones: Interactions Between Socioeconomic and Natural Systems. Environmental Management 20: 159-73.

Vitousek, P. M., Mooney, H. A., Lubchenco, J., Melillo, J. M. 1997. Human Domination of Earth's Ecosystems. Science 227: 494-9.

Wilson, M. A., Costanza, R., Boumans, R. M. J., Liu, S. L. 2005. Intergrated Assessment and Valuation of Ecosystem Goods and Services Provided by Coastal Systems. In *The Intertidal Ecosystem*, ed. (James G. Wilson),1-24 pp. Royal Irish Academy Press. Dublin, Ireland.

Wilson, M. A., Hoehn, J. P., eds. 2006. Special Issue--Environmental Benefits Transfer: Methods, Applications and New Directions. *Ecological Economics*. Forthcoming.

Wilson, M. A., Troy, A., Costanza, R. 2004. The Economic Geography of Ecosystem Goods and Services: Revealing the monetary value of landscapes through transfer methods and Geographic Information Systems. In *Cultural Landscapes and Land Use*, ed. (M. Dietrich, Van Der Straaten). Kluwer Academic.

A meta-analysis of contingent valuation studies in coastal and near-shore marine ecosystems

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ABSTRACT

The ecosystem services provided by coastal and nearshore marine systems contribute significantly to human welfare. However, studies that document values of these services are widely scattered in the peer-reviewed literature. We collected 39 contingent valuation papers with 120 observations to conduct the first meta-analysis of the ecosystem service values provided by the coastal and nearshore marine systems. Our result showed over ¾ of the variation in Willingness to Pay (WTP) for coastal ecosystem services could be explained by variables in commodity, methodology, and study quality. We also used the meta-regression model to predict out-of-sample WTPs and the benefit transfer result showed that the overall average transfer error was 24%, with 40% of the sample having transfer errors of 10% or less, and only 2.5% of predictions having transfer errors over 100%. Based on such results, one could argue that such meta-analyses can provide useful guidance regarding at least the general magnitudes of welfare effects. However we also caution against the application of such a result in a broader context of benefit transfer as it is derived from a limited amount of data, and it may suffer from some degree of measurement error, generalization error, and publication selection error. Lastly, we discussed the sources of these errors and future research plans concerning how to minimize them.

KEY WORDS Meta-analysis, Ecosystem services, Contingent valuation, Benefit transfer, Coastal ecosystems

Meta-analysis has been applied extensively in fields such as education and medical sciences where applications involve studies conducted under controlled conditions with standardized experimental designs (van den Bergh et al. 1997). However, it is still used sparingly in ecosystem service valuation because of the heterogeneity of research methods in economics and a lack of standardized data reporting.

Within a benefit transfer context, meta-analysis can provide information to allow researchers to more appropriately adjust benefit estimates. Based on this potential, USEPA guidelines characterize meta-analyses as “the most rigorous benefit transfer exercises” (p. 87) (EPA 2000)

The purpose of this study is to 1) assess whether variation in WTP for coastal ecosystem services may be explained sufficiently by systematic variation in contextual variables to justify benefit transfer, 2) use the meta-regression model for out-of-sample benefit transfer and calculate the transfer error, and 3) discuss the sources for the transfer errors and how to minimize them in future research.

Meta-analysis and function transfer

Gene V. Glass published his ground breaking article on Meta Analysis (MA) in 1976. In that article, he laid out the fundamental rationale for the technique and defined many of the basic features of MA as it is known and used today. He also coined the term “meta-analysis”, which he defined as:

> “…the statistical analysis of a large collection of results from individual studies for the purpose of integrating the findings. It connotes a rigorous alternative to

the casual, narrative discussions of research studies which typify our attempts to make sense of the rapidly expanding research literature (Glass 1976, p3)".

The concept of meta-analysis has a considerable history in the natural sciences, but only recently has it begun to influence the social sciences in general and economics specifically (van den Bergh et al. 1997). In the field of environmental economics, *Meta-analysis* refers specifically to the practice of using a collection of formal and informal statistical methods to synthesize the results found in a well-defined class of empirical studies (Smith and Pattanayak 2002). MA has three general purposes: 1) synthesize past research on a particular topic, 2) test hypotheses with respect to the effects of explanatory variables, and 3) use the meta-regression model in function transfer (Bergstrom and Taylor 2006). Traditionally MA has been used for the first two purposes but a more recent use is the systematic utilization of the existing value estimates from the source literature for the purpose of benefit transfer (e.g. Rosenberger and Loomis 2000, Johnston, et al. 2005, Brander et al. 2006).

The first two meta-analyses in the field were by Walsh and colleagues on outdoor recreation benefit and by Smith and Kaoru on travel cost studies of recreation benefits in the late 1980s and early 1990s (Walsh et al. 1989, Walsh et al. 1992, Smith and Kaoru 1990). More recent applications of MA for similar purposes include groundwater (Boyle et al. 1994), air quality and associated health effects (Smith and Huang 1995, Desvousges et al. 1998), endangered species (Loomis and White 1996), air pollution and visibility (Smith and Osborne 1996), and wetlands (Brouwer et al. 1997, Brouwer et al. 1999, Woodward and Wui 2001).

In the context of benefit transfer, meta-analysis enables us to statistically explain the variation found across empirical studies. Once the basic model specification is complete, that is, if it includes the relevant explanatory variables in the correct functional form, then the net benefit estimate for the policy site can be estimated by inserting values of explanatory variables into the function (Walsh et al. 1992). Of course, the basic premise is the existence of an underlying valuation function.

Meta-analysis has two major conceptual advantages over other value transfer approaches such as point estimate and demand function transfers (Rosenberger and Loomis 2000, Shrestha and Loomis 2003):

1) Meta-analysis utilizes information from a greater number of studies, thus providing more rigorous measures of central tendency that are sensitive to the underlying distribution of the study site measures.
2) Methodological differences between different non-market valuation techniques can be controlled when calculating a unique value estimate from the meta-analysis function.

Based on this potential, USEPA guidelines characterize meta-analyses as "the most rigorous benefit transfer exercises" (p. 87) (EPA 2000). On the other hand many limitations of benefit transfers in general are also applicable to meta-analysis (Desvousgese et al. 1998). These are briefly listed below:

1) There should be sufficient original studies conducted so that statistical inferences can be made and relationships modeled.
2) A meta-analysis can only be as good as the quality of the research that is included. This includes the scientific soundness of the original research and the transparency in reporting results and summary statistics for the original data.
3) The studies included in the analysis should be similar enough in content and context that they can be combined and statistically analyzed.

In sum, the use of meta-analysis in value transfer is fairly new and very promising but it is not without its limitations. First and foremost, it depends heavily on the quality of the primary studies used. As the quality of information increases within the source literature, the accuracy of the resulting meta-analysis technique will likely improve.

METHOD

Data selection

Empirical valuation data is often scattered throughout the scientific literature and is uneven in quality. We selected studies that deal *explicitly* with non-market coastal ecosystem services measured throughout the world and focused on peer-reviewed ones only because of their presumably higher quality. Our literature review yielded a total of 70 studies and most of them featured the contingent valuation (CV) technique (Wilson and Liu 2007). Therefore, we selected this subset of studies for further analysis.

Only 39 of these studies reported benefit estimates or provided sufficient information to derive them. From these 39 studies we coded 120 observations for our meta-analysis. Several studies are responsible for multiple observations because they reported alternative results due to the use of split survey samples targeting different groups and/or testing different survey designs[1]. Care was taken not to double count benefit estimates reported by the same authors in more than one paper.

Data coding

Based on the theory and findings in the literature, we expect that various attributes may be associated with systematic variations in WTP for coastal ecosystem services. Following Bergstrom and Taylor (2006) these attributes are categorized into those characterizing 1) commodity consistency, 2) methodology consistency, and 3) data quality consistency between study and policy sites. Commodity attributes characterize the subjects (i.e. income and density of the surveyed population), object (e.g. ecosystem services type and land cover type), and marginal change in the valuation (type and degree of the change).

Table 1 summarizes this set of 50 independent variables. The majority are qualitative dummy variables coded as 0 or 1, where 0 means the study does not have that characteristic and 1 means that it does. One of the biggest limitations of meta-analysis is the lack of comparability across studies (Woodward and Wui 2001). Characteristics of valuation are often reported in such a diverse manner that the best a meta-analyst can do is to use a binary variable to indicate whether an attribute is associated with each observation.

[1] We coded all value estimates reported in a single study , which exposes the dataset to the danger of selection bias as estimates from the same study were likely more similar.

Sometimes these explanatory variables were not explicitly reported at all in the source papers because they define the context of the valuation, and therefore, were treated as constants in the original studies. As a result external sources have to be used to extract such information. In particular, income data for the survey respondents is not reported in most cases. In these cases we used the mean GDP per capita adjusted for purchasing power parity (PPP) (Penn World Table, http://pwt.econ.upenn.edu/php_site/pwt_index.php) in the country in which the surveyed sample resides to account for people's capacity to pay. For the U.S. studies, regional income information was gathered from the US Department of Commerce's online database (http://bea.gov/regional/index.htm#state).

Survey year was adopted as a surrogate for quality of a valuation study. Another possible indicator of quality is the survey response rate, but about one quarter of our studies did not report this, and in those studies that did report it is often unclear what these response rates actually represent or which criteria may have been used to exclude responses from further analysis (Brouwer et al. 1999).

All of the WTP measures were converted to 2006 USD dollars (by using the Consumer Price Index) per household per year. We created the binary variable "*Whether primary data only"* to identify those studies that gave enough information for the conversion. 0 means external sources were used to during the conversion.

Model construction

Meta-analyses have utilized a range of statistical models including *Ordinary Least Squares* (OLS) (e.g. Rosenberger and Loomis 2000, Schlapfer 2006, and Brander et al. 2006) and the *multilevel model* (e.g. Bateman and Jones 2003 and Johnston et al. 2005), leaving researchers to make *ad hoc* judgments regarding the most appropriate statistical specification for meta-models.

We used OLS and a nonlinear Box-Cox procedure to estimate our model[2]. We estimated a number of OLS regressions with different functional forms to search for a model with residuals with desirable properties. These included a linear model, a model with a logarithmic dependent variable, a model where the continuous explanatory variables were in logarithms but the dependent variable was not, and a log-log model (the qualitative variables were not transformed in any of these specifications). We also tried a fairly general specification search using Box-Cox transformations for the continuous variables. This showed that the Box-Cox parameter was not significantly different from zero and, therefore, the model could be approximated by a log-log model. In order to test if omitting irrelevant variables might help reduce multi-collinearity we then applied a stepwise regression procedure to the log-log model by stepping out variables.

The general model is:

$$f(y_i) = \alpha + \sum_j \beta_j g(x_{ji}) + \sum_k \gamma_k z_{ki} + \varepsilon_i$$

Where f () and g () are the functions used to transform the dependent variable y and continuous explanatory variables x respectively. z are the qualitative explanatory variables (dummies) and ε is the error term. α, β_j, and γ_k are regression coefficients and individual observations are indexed by i.

[2] A multi-level model was considered but not adopted. This approach allows for the often unrealistic assumption of independence between estimates to be relaxed by using dummy variables for each group within each level (e.g. study sites, author, method and study). But this approach is only feasible when the data set is homogenous or there are a large number of observations available to run the model. Unfortunately neither is the case for our dataset.

Function transfer

Following Brander et al. (2006) we predicted the WTP for each of the 120 observations by using the value transfer function estimated on the other 119 observations. Then we compared the predicted WTP to the "actual" WTP in the original study to calculate the transfer error, defined as $| (WTP_{act}-WTP_{prd}) / WTP_{act}|$.

[Insert Table 1]

RESULTS

Summary statistics

The average annual per household WTP is about $766 (USD2006). The median however is $88.5 per household per year, showing that the distribution is skewed with a tail of high values. As expected, the mean WTP varies considerably depending on the coastal ecosystem services considered, the land cover, study area, and valuation method. Table 2 presents the breakdown of WTPs by 1) ecosystem service, 2) land cover, 3) geopolitical region, and 4) CV elicitation method. The information here does not account for interaction between explanatory variables. We use meta-regression in the next section to examine the importance of each variable in explaining the variation in WTP while accounting for variation in the other variables.

The wide range of WTP values by ecosystem service is striking though not unexpected for coastal ecosystems (Costanza et al. 1997, Costanza et al. 2007). Average annual per household willingness to pay ranges from $0.30 for provisioning of food and $1.5 for disturbance control to $3,268 for aesthetic services. It is worthwhile to notice that we only have one observation for both food and disturbance services, and the Standard Deviation (SD) of aesthetic services is quite high as well.

In terms of land cover type, saltwater wetland, marsh, or pond has the highest average WTP of \$2189 household^{-1} year^{-1}(again with a high SD), and the near-shore islands and beaches have values at the lower end of the spectrum (\$37 and \$38 US \$ household^{-1} year^{-1}, respectively). Compared to a recent study (Costanza et al. 2007) where the total ecosystem service value of beaches in the State of New Jersey was estimated as \$42,147 acre^{-1} year^{-1}(USD 2004), this beach value seems surprisingly low. This latter value is the value of an acre of beach aggregated across all relevant households, while the value in the current study is the WTP of a single household.

Average WTP values are highest in North America, followed by Asia, Oceania, South America, and Europe although 75% of our data points refer to North America. The geographical distribution of observations in our sample reflects the availability of valuation studies rather than the distribution of coastal and near-shore marine ecosystems.

In comparison, grouped by elicitation format the dataset has a much more even distribution. Studies using the contingent ranking produce the highest values, followed by those using contingent behavior (including both contingent behavior and combined CV and RP studies), and dichotomous choice. On the other end of the spectrum, iterative bidding studies have the lowest WTP values. These results are in line with the literature, as it is well known that different ways of asking preference questions yield different estimates of willingness to pay (e.g. Desvousges et al. 1987). Open-ended, payment card, and iterative bidding approaches are all believed to open the possibility of free-riding, therefore leading to an understatement of WTP (Bateman and Jones 2003). On the other hand, WTP value estimates from a contingent ranking exercise have been recently found to be greater than those elicited through CV (Stevens et al. 2000, Bateman et al. 2006).

[Insert Table 2]

Meta-regression

We estimated a number of regressions with different functional forms to see if we could find a model with residuals having desirable properties. Table 3 presents coefficients, significance level (for the continuous variables only for the sake of brevity), and results of diagnostic tests for each model.

First we estimated a regression where all variables enter linearly. The last variable in each group of dummies was dropped from the regression to avoid collinearity (marked with an asterisk in Table 1). The standard errors were estimated using the ROBUSTERRORS option in the RATS (Regression Analysis for Time Series) econometrics package so that the standard errors of the coefficients would take into account for potential heteroskedasticity of unknown form. Income and survey year are non-significant and both even have the wrong sign. Density is significant but unexpectedly has a negative sign. Area of the study has the expected result. The residuals have very strong kurtosis (4^{th} moment = fat tails) though skewness is not significant. Therefore, the Jarque-Bera normality test rejects the null that the residuals are normally distributed. The Breusch-Pagan heteroskedasticity test checks the correlation between the squared residuals and the full set of explanatory variables. It strongly rejects the null of homoskedasticity.

Next we tried a fairly general specification search applying a Box-Cox transformation to the dependent variable and the continuous explanatory variables (RATS Manual, 280).[3] We estimated the models using maximum likelihood. The result showed the value of λ is not significantly different from zero, which indicates that the model is close to log-log. All the key continuous explanatory variables have positive and highly significant coefficients. The residuals are now homoskedastic but skewness and kurtosis have deteriorated.

The third model we present is a log-log model where both dependent variable and continuous independent variables are transformed into natural logarithms. The coefficients of the continuous variables have the expected sign but only that of area is significantly different from zero. Though there is no heteroskedasticity the residuals are highly non-normal.

In order to see if omitting irrelevant variables might help reduce multi-collinearity we optimized the model by retaining only those variables that were significant at a 20% level of confidence or better based on t-statistics using the STWISE procedure in RATS. The procedure started with the full vector of explanatory variables and “stepped out” non-significant variables. We estimated this final model using the ROBUSTERRORS option for the standard errors of the regression coefficients. As expected compared to the log-log model the corrected R-squared increases. The t-statistics also increase a little to

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[3] The Box-Cox transformation f(x) is given by: $f(x) = \frac{x^{\lambda} - 1}{\lambda}$ where λ is a parameter to be estimated. This function is nonlinear in the parameters and therefore λ cannot be estimated by OLS. When the dependent variable is also subject to Box-Cox transformation an explicit maximum likelihood estimation procedure is required (RATS Manual, 280).

be somewhat more significant. The residual properties are slightly better than the full model as well but are still non-normal.

[Insert Table 3 and Table 4]

Table 4 lists the coefficients and significance levels of all the explanatory variables of the step-wise model. The R^2 for this model is 0.79, indicating that approximately 4/5 of the variation in WTP is systematically explained with model variables. Furthermore, the signs of the significant parameter generally correspond with intuition, where prior expectations exist.

For the dummy variables the coefficients indicate the percentage change in the dependent variable for the presence of the characteristic indicated by the dummy variable relative to the value of the dependent variable in the base case. For the continuous variables, the coefficients should be interpreted as elasticities, that is, the percentage change in the dependent variable given a small percentage change in the explanatory variable.

<u>Commodity consistency: the subject of the valuation</u>

Coefficients on the *income* of survey respondents *and population density* are both positive, and the former is significant at 6% and latter only at the 16% level. The coefficient for income is 0.42, suggesting a 10% increase in income leads to roughly a 4% increase in WTP for coastal ecosystem services. This finding echoes the usual empirical result from CV studies where a positive income elasticity of WTP was found to be substantially less than one for environmental commodities (Kristrom and Riera 1996, Carson et al. 2001, Horowitz and McConnell 2003).

Commodity consistency: the object of the valuation

Compared to the baseline service of water supply, the WTPs for *food* provision and for *spiritual* service are both significantly lower (p=0.0000 and 0.078, respectively). This corresponds with past meta-analysis where the value of provisioning service and non-use value were found to be small (Brander et al. 2006, Johnston et al. 2005). However the first part of the result has to be interpreted with caution because there is only one observation for food service in our dataset.

Separation of direct, indirect use and non-use benefit is difficult sometimes. Brouwer et al. found only in a third of all CV studies could a single benefit flow be identified, in all other cases wetlands provided multiple benefits (1999). In order to take account of this effect we created a dummy variable of *Bundled service* to investigate whether it can explain variations of WTP. The coefficient turned out to be negative and significant at an 11.2% level, which makes intuitive sense because a package of goods should be valued less than the sum of its independently valued constituents.

The coefficient on the size of the study *area* is positive and very significant and a coefficient of 0.17 indicates that doubling of the study area size will only lead to a 17% increase in lnWTP, which signals decreasing returns to scale as documented in the past research (Woodward and Wui 2001, Brander et al. 2006).

Compared to the baseline of *Asia* as the study location, people seemed to be more willing to pay for coastal ecosystem services in *Europe* but less so in the *Oceanic* area (both significant at 5% level). The coefficient for South America is also positive and significant but given the paucity of observations (n=1), it is possible that the significance of the coefficient is entirely due to a single study and has nothing to do with a fundamental difference.

WTPs for beach, estuary, and open ocean are lower than that of the semi-enclosed sea (baseline). Again the beach value is surprisingly low, compared to the result of our recent study (Costanza et al. 2007) where the total ecosystem service value of beach in the State of New Jersey was estimated as the highest among coastal and marine systems (other land cover valued include coastal shelf, estuary and saltwater wetland). The difference could be perhaps explained by the different units of valuation.

<u>Commodity consistency: variables of marginal change</u>

The default category here is a negative change in the service. Compared to this baseline, lower WTP is associated with *no change, 100% and 200% positive changes*[4]. Furthermore, the coefficients showed that WTP is higher for 100% positive change than for 200% change, which indicates WTP is sensitive to the scope of improvement but only to some extent. Indeed for many environmental goods the public may have sharply declining marginal utility after a reasonable amount of it has been provided (Rollins and Lyke 1998).

<u>Methodology consistency</u>

The contingent ranking (CR) is used as the baseline category in the regression analysis in order to avoid collinearity. The negative coefficients for the other five *elicitation formats* indicate that these formats generate lower WTP values than the

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[4] Because it is impossible to compare changes over different ecosystem services studies, the changes here are relative compared to their own baseline of *status quo*. For instance, for water quality studies, a 100% water quality improvement means moving up a step along the water quality ladder. For recreation fishing studies this means 100% increase of fish population.

baseline (all highly significant). Corresponding to previous research results, other elicitation formats produced significantly lower WTP than contingent ranking (Stevens et al. 2000, Bateman et al. 2006). Stevens et al. (2000) provide three reasons why CR and CV results may differ. 1) substitutes are often made more explicit in the ranking format and. therefore, respondents are encouraged to explore their preferences and trade-offs in greater depth, 2) the psychological process of ranking in the CR format is somewhat different than that of the CV format, 3) non-response and protest zero-bidding behavior may be less of a problem for CR because it is easier to express indifference to the choices by ranking them equally.

Among different CV elicitation formats, the results also corresponds to past empirical research conclusions that WTP estimates from binary discrete choice formats tend to be higher than those from other formats (Boyle et al. 1994, Carson et al. 2001).

Interview (including both face to face and phone interview) has a negative and statistically significant coefficient ($p < 0.05$) compared to the default of mail surveys. This finding contradicts the previous empirical evidence where “warm glow” has been offered as a possible explanation why interview-based WTP might be higher. Respondents in a face-to-face CV survey may attempt to please an interviewer by agreeing to pay some amount when they would not do so otherwise (Carson et al. 2001).

However, our contradictory result may be because we pooled together face-to-face with phone interview studies. In the future they should be separated and at least one other meta-analysis show that both face-to-face interviews and mail surveys have positive and significant coefficients in comparison to telephone surveys (Johnston et al. 2005).

The coefficient estimated for the dummy variable '*payment vehicle*' reflects, *ceteris paribus*, an almost 30% higher average WTP for an increase in tax than the baseline payment type of donation (p=.107). This result is comparable to that of Brower et al (1999), where the difference was about two times larger. One possible explanation is that to use taxation as a payment vehicle is expected to prompt responses which consider the benefits for society at large and not just restricted to private use only. Another way to explain it is that the unwillingness among respondents to offer large voluntary payments is due to their fear that others will ride for free.

WTP values for the majority of studies included in the analysis consist of annual payments over an indefinite duration. However, a small number of studies estimate WTP for one-time payments. The variable *lumpsum* identifies studies in which payments were to occur other than on an annual basis. The positive and statistically significant parameter for lumpsum reveals sensitivity to the payment schedule. Studies that ask respondents to report an annual payment (as opposed to a shorter lumpsum payment) have lower nominal WTP estimates ($p < 0.01$).

The variable of *Sub-sample* was used to investigate the influence of dropping outliers when calculating the central tendency of WTP in the CV studies. As expected, smaller WTP estimates are associated with studies that eliminate or trim outlier bids ($p<0.05$).

Variables on study quality

Without a better choice, *Survey Year* was adopted as an indicator for quality of the study (Johnston et al. 2005). The premise is that as the focus of stated preference survey design improves over time, there has been a reduction of survey biases that would otherwise result in an overstatement of WTP. The negative sign of the coefficient means that later studies are associated with lower WTP (p=0.036).

However, this variable may also represent whether ecosystem services are growing more or less scarce over time. Unfortunately, the influence of systematic refinements in methodology over time cannot be distinguished from a scarcity-related trend in the availability of ecosystem services relative to demand (Smith and Kaoru 1990).

Function transfer

Figure 1 plots the "actual" and predicted natural log values of the dependent variable and Figure 2 showed the transfer error associated with each observation ranked in order of ascending WTP.

The overall average transfer error is 24%, ranging from 0% to 430%. In comparison to other function transfer exercises, our results appear to be similar despite the relative diversity of our data (see summary table of transfer validity tests in Rosenberger and Stanley, 2006).

The average transfer error for different quartiles of the data series ordered by "actual" WTPs in ascending order is 56%, 18%, 12% and 10%, respectively, with 40% of the sample having transfer errors of 10% or less. Only 2.5% or 3 out of the 120 predictions resulted in transfer errors over 100%, and these 3 are associated with the three lowest WTPs. This indicates that the fit for low ecosystem service values is poor compared to medium to high values.

These large errors could probably be related to the low incidence of specific characteristics associated with these three data-points. In other words, their attributes are under-represented in our meta-database. The observation with the highest transfer error, for instance, is from a study on food provision service, for which we have only one data point. Indeed, if we view each empirical study included in the meta-analysis as a sample of this meta-function, then this function becomes an envelope of study site functions that relate WTP and the context variables. If some variables of the policy site are outside this envelope to start with, then one can predict a large transfer error.

Essentially this is the type of generalization error discussed by Rosenberger and Stanley (2006). It arises when estimates from study sites are adapted to represent policy sites with different conditions. These errors are inversely related to the degree of similarity between the study and the policy site. Rosenberger and Stanley also discussed another two general types of errors in benefit transfer: measurement and publication bias errors. Measurement error occurs when a researcher's decisions affect the accuracy of the transferability, publication bias error happens when the empirical literature included in the meta-analysis is not an unbiased sample of empirical evidence. They both relate to issues in ecosystem service valuation in general and will be covered in detail in the next section.

[Insert Figure 1 and Figure 2]

DISCUSSION

Measurement error: more than a problem of original studies

Measurement error stems from the judgments and the methods used in the original study. During meta-analysis, a portion of measurement error will be 'passed through' if effort is not taken to minimize it (Wilson and Cohen 2006). Put another way, the accuracy of benefit transfer is subject to the measurement of original studies and in fact some have argued, "Benefit transfers can only be as accurate as the initial benefit estimates (Brookshire and Neill 1992)."

Fifteen dummy variables were used in order to maintain methodological consistency in our model and 9 of them turned out to be significant in the step-wise model. However, there are a couple of limitations in this approach: 1) any model estimated using a large number of dummies will quickly become large and complex therefore the degree of freedom and the explanatory power of the model will decrease. In this case one has to somehow pool dummy variables in a meaningful way. The effort in combing through face-to-face and phone interviews was such an attempt. 2) Critical information needed for data-coding is missing from the original studies.

This problem of incomplete information is not only restricted to methodology related variables. Brouwer et al. found in their meta-analysis research that two-thirds of their original studies contained no information about the size of the area involved (1999). This is rather unfortunate considering, along with other researchers (e.g. Woodward and Wu 2001, Brander et al 2006), we found that the size of the study area has a significant explanatory power for WTP variations.

When no information is readily available from the original study, meta-analysis researchers are forced to use external sources during their data coding process. For instance another category of information often missing is user population details. In the most comprehensive benefit transfer exercise on recreational service, out of the 131 studies included about 3% of the studies reported average income for their samples, less than 1% reported average education level, about 16% reported gender proportions, and only 61% bothered to report their sample size (Rosenberger and Stanley 2006). Rosenberger and Loomis (2000) did attempt to proxy user population characteristics by using U.S. Census data for the state in which the study was conducted, but found in preliminary analysis that these proxies were broadly insensitive to differences in benefit measures provided.

When there is even no proxy available a "N vs. K' dilemma is posed: should the researcher discard explanatory variables that are not common to all studies (thus preserve N at the cost of K) or discard observations that do not include key regressors (thus preserve K at the cost of N) (Moeltner et al. 2007)? This is a difficult question and it is every researcher's judgment call.

We attempted to maintain a balance between the two. We resort to external resources for income, population density, and the size of study area to preserve N. On the other hand in order to preserve K we didn't delete those variables with only one observation including Food provision service, disturbance control service, and the study with South America as its study site. It is likely any other idiosyncratic factors that affect a single observation may be attributed spuriously to that characteristic. In this sense the measurement error is not only due to the original research but could also come from the meta-analysis process itself.

In addition to the of use dummy variables, another way to minimize the measurement error is controlling the quality of the original studies used in the meta-analysis. This was done by selecting studies from peer-reviewed papers only. Johnston et al. (2005) did it as well by focusing on those studies with methods "generally accepted by journal literature (p223)[5]". It could well be a coincidence but the regression models from these two studies both have adjusted R^2 higher than 0.75.

Though it is possible that quality control means a meta-model with a higher explanatory power, the cost of doing so is to expose researchers to selection bias error.

Publication selection Bias: how to avoid the inevitable?

Publication selection bias, or the 'file drawer problem', has been a major concern for using meta-analysis in economics (Stanley 2001, Stanley 2005). A sample of value estimates that approximates a random draw is assumed, but this assumption is unlikely to be met because meta-data are often subject to various forms of selection bias. For instance, researchers and reviewers are predisposed to treat statistically significant results more favorably and as a result they are more likely to be published. Studies that find relatively 'non-significant' effects tend to left in the 'file drawer'.

For this reason meta-analysts are encouraged to mitigate the selection bias by including grey literature and any unpublished reports they can find. "It is best to err on the side of inclusion," as Stanley put it (2001). Next, statistical methods can be employed to identify and/or accommodate these biases (Stanley 2005, Hoehn 2006).

Several recent economic meta-analyses attempted to overcome this problem by including an extra dummy variable that identifies the publication type (whether peer-reviewed or not). Woodward and Wui (2001) did not find a significant effect from

[5] Their selection included non-peer reviewed literature as well. This paper did not adopt their approach because to decide what is "acceptable for journal literature" meant another layer of subjective judgment, which was to be avoided as much as possible.

publication type in explaining variation of their wetland WTP data. But Rosenberger and Loomis (2000) showed that not only do journal publications have a smaller aggregate mean estimate than non-journal publications, but there is also greater variation in estimates provided across published studies.

One possible explanation is the accuracy of the reported estimates in the peer-reviewed literature may be less than ideal (Rosenberger and Stanley 2006). This is because most journals are not interested in publishing new estimates for their own sake and the current institutional incentives have criteria biased toward methodological and theoretical contributions (Smith and Pattanayak 2002). In this sense publication selection bias is more a matter of methodological innovation than statistical significance in the area of ecosystem service valuation (ESV) (Loomis and Rosenberger 2006).

Another layer of selection bias in the ESV field is due to funding availability. Valuation research is costly and such costs limit the feasibility of many original studies (though it also promotes benefit transfer). Decisions to fund research are linked to human awareness of the importance of ecosystem services and the magnitude of the policy decisions made in response to conflicts over resource use (Hoehn 2006). Such decisions are certainly not random. As Woodward and Wu noticed (2001) wetlands that are considered valuable *a priori* are much more likely to be valued. On the other hand, our results show that *Marquee Status* was not significant in the step-wise model.

Although selection bias does not necessarily lead to errors in estimation of the valuation function, given the limitations of available data, the likelihood of such bias should be taken into account in future benefit transfer exercises. What is particularly important is to avoid measurement error and publication selection bias working in the same direction.

In summary, it seems difficult to avoid selection bias as it is more of a 'system error' at macro-level. On the other hand, there are methods available to minimize it at micro-level. In the next section the possible selection bias of our dataset will be discussed, and then a plan sketched for future research.

DIRECTION FOR THE FUTURE

As mentioned separately in previous sections, the values in our data are also not independent draws for a couple reasons: 1) it has panel characteristics because some studies and authors generate multiple WTP estimates (Smith and Kaoru 1990), and 2) it includes peer-reviewed literature only.

There have been two ways to deal with the issue of panel data in literature: to use corrective procedures (Smith and Kaoru 1990, Rosenberger and Loomis 2000), or to statistically check and test for, and model this potential panel effect (Brouwer et al. 1999, Bateman and Jones 2003, Johnston et al. 2005). In this study it was decided to adopt a corrective procedure by using the ROBUSTERROR option to correct the standard errors of the regression coefficients for potential heteroskedasticity. But this still does not account for common effects due to several studies or WTP estimates being produced by a single author or group of authors. Therefore, *one potential future direction* is to statistically test for these effects by using a panel data model or multi-level model. A daunting challenge of the former though, is to identify the possible source of these effects because sources of heterogeneity and correlation may not be based on a single dimension such as study and researcher. A multi-level model requires a much larger and/or more homogeneous dataset, which is unavailable.

Therefore, *the natural next step* is to enlarge the dataset by adding non-peer reviewed literature. Another bonus of doing so would be to minimize publication selection barriers. We could also introduce one dummy variable indicating whether a study is peer-reviewed or not, in order to test the effect of selection bias.

REFERENCE

Bateman, I. J., M. A. Cole, et al. (2006). "Comparing contingent valuation and contingent ranking: A case study considering the benefits of urban river water quality improvements." Journal of Environmental Management **79**(3): 221-231.

Bateman, I. J. and A. P. Jones (2003). "Contrasting conventional with multi-level modeling approaches to meta-analysis: Expectation consistency in UK woodland recreation values." Land Economics **79**(2): 235-258.

Bergstrom, J. C. and L. O. Taylor (2006). "Using meta-analysis for benefits transfer: Theory and practice." Ecological Economics **60**(2): 351-360.

Bergstrom, J. C. and L. O. Taylor (2006). "Using meta-analysis for benefits transfer: Theory and practice." Ecological Economics **60**(2): 351-360.

Boyle, K. J., G. L. Poe, et al. (1994). "What Do We Know About Groundwater Values - Preliminary Implications from a Meta-analysis of Contingent-Valuation Studies." American Journal of Agricultural Economics **76**(5): 1055-1061.

Brander, L. M., R. Florax, et al. (2006). "The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature." Environmental & Resource Economics **33**(2): 223-250.

Brookshire, D. S. and H. R. Neill (1992). "Benefit Transfers - Conceptual and Empirical Issues." Water Resources Research **28**(3): 651-655.

Brouwer, R., I. H. Langford, et al. (1997). A meta-analysis of wetland contingent valuation studies. Norwich, CSERGE, University of East Anglia.

Brouwer, R. and F. A. Spaninks (1999). "The validity of environmental benefits transfer:

Further empirical testing." Environmental & Resource Economics **14**(1): 95-117.

Carson, R. T., N. E. Flores, et al. (2001). "Contingent valuation: Controversies and evidence." Environmental & Resource Economics **19**(2): 173-210.

Costanza, R., R. dArge, et al. (1997). "The value of the world's ecosystem services and natural capital." Nature **387**(6630): 253-260.

Costanza, R., M. Wilson, et al. (2007). The Value of New Jersey's Ecosystem Services and Natural Capital, New Jersey Department of Environmental Protection.

Desvousges, W. H., F. R. Johnson, et al. (1998). Environmental policy analysis with limited information: principles and application of the transfer method., Edward Elgar.

Desvousges, W. H., V. K. Smith, et al. (1987). "Option price estimates for water quality improvements: A contingent valuation study for the Monongahela river." Journal of Environmental Economics and Management **14**(3): 248-267.

EPA, U. S. (2000). Guidelines for Preparing Economic Analyses. EPA 240-R-00-003. . Washington, D.C., U.S. Environmental Protection Agency.

Glass, G. V. (1976). "Primary, secondary, and meta-analysis of research." Educational Researcher **5**: 3-8.

Hoehn, J. P. (2006). "Methods to address selection effects in the meta regression and transfer of ecosystem values." Ecological Economics **60**(2): 389-398.

Horowitz, J. K. and K. E. McConnell (2003). "Willingness to accept, willingness to pay and the income effect." Journal of Economic Behavior & Organization **51**(4): 537-545.

Johnston, R. J., E. Y. Besedin, et al. (2005). "Systematic variation in willingness to pay for aquatic resource improvements and implications for benefit transfer: a meta-analysis." Canadian Journal of Agricultural Economics-Revue Canadienne d' Agroeconomie **53**(2-3): 221-248.

Kristrom, B. and P. Riera (1996). "Is the income elasticity of environmental improvements less than one?" Environmental and Resource Economics **7**(1): 45-55.

Loomis, J. B. and R. S. Rosenberger (2006). "Reducing barriers in future benefit transfers: Needed improvements in primary study design and reporting." Ecological Economics **60**(2): 343-350.

Loomis, J. B. and D. S. White (1996). "Economic benefits of rare and endangered species: Summary and meta-analysis." Ecological Economics **18**(3): 197-206.

Moeltner, K., K. J. Boyle, et al. (2007). "Meta-analysis and benefit transfer for resource valuation-addressing classical challenges with Bayesian modeling." Journal of Environmental Economics and Management **53**(2): 250-269.

Rollins, K. and A. Lyke (1998). "The Case for Diminishing Marginal Existence Values." Journal of Environmental Economics and Management **36**(3): 324-344.

Rosenberger, R. S. and J. B. Loomis (2000). Benefit transfer of ourdoor recreation use values, USDA Forest Service.

Rosenberger, R. S. and T. D. Stanley (2006). "Measurement, generalization, and publication: Sources of error in benefit transfers and their management." Ecological Economics **60**(2): 372-378.

Schlapfer, F. (2006). "Survey protocol and income effects in the contingent valuation of public goods: A meta-analysis." Ecological Economics **57**(3): 415-429.

Shrestha, R. K. and J. B. Loomis (2003). "Meta-analytic benefit transfer of outdoor recreation economic values: Testing out-of-sample convergent validity." Environmental & Resource Economics **25**(1): 79-100.

Smith, V. K. and Y. Kaoru (1990). "Signals or Noise - Explaining the Variation in Recreation Benefit Estimates." American Journal of Agricultural Economics **72**(2): 419-433.

Smith, V. K. and L. L. Osborne (1996). "Do contingent valuation estimates pass a "scope" test? A meta-analysis." Journal of Environmental Economics and Management **31**(3): 287-301.

Smith, V. K. and S. K. Pattanayak (2002). "Is meta-analysis a Noah's ark for non-market valuation?" Environmental & Resource Economics **22**(1-2): 271-296.

Stanley, T. D. (2001). "Wheat from chaff: Meta-analysis as quantitative literature review." Journal of Economic Perspectives **15**(3): 131-150.

Stanley, T. D. (2005). "Beyond publication bias." Journal of Economic Surveys **19**(3): 309-345.

Stevens, T. H., R. Belkner, et al. (2000). "Comparison of contingent valuation and conjoint analysis in ecosystem management." Ecological Economics **32**(1): 63-74.

van den Bergh, J. C. J. M., K. Button, et al. (1997). Meta-analysis in environmental economics, Kluwer Academic Publishers.

Walsh, R. G., D. M. Johnson, et al. (1989). "Issues in nonmarket valuation and policy application: a retrospective glance." Western Journal of Agricultural Economics **14**: 178-188.

Walsh, R. G., D. M. Johnson, et al. (1992). "Benefit Transfer of Outdoor Recreation

Demand Studies, 1968- 1988." Water Resources Research **28**(3): 707-713.

Wilson, M. and S. Liu (2007). Evaluating the Non-Market Value of Ecosystem Goods and Services provided by Coastal and Nearshore Marine System. . Ecological Economics of the Oceans and Coasts. . G. Patterson and Glavovic.

Wilson, M. A. and J. P. Hoehn (2006). "Valuing environmental goods and services using benefit transfer: The state-of-the art and science." Ecological Economics **60**(2): 335-342.

Woodward, R. T. and Y. S. Wui (2001). "The economic value of wetland services: a meta-analysis." Ecological Economics **37**(257-270).

Table 1: Explanatory variables of meta-analysis

Variable	Description	Data type
Commodity consistency		
--Objects of valuation		
(Ecosystem services)		
BUNDLED_ES	Multiple services	Binary (0 or 1)
ES_AES	Aesthetic service	Binary (0 or 1)
ES_DIS	Disturbance control	Binary (0 or 1)
ES_FOOD	Food	Binary (0 or 1)
ES_HAB	Habitat	Binary (0 or 1)
ES_REC	Recreation	Binary (0 or 1)
ES_SPR	Spiritual	Binary (0 or 1)
ES_WAS*	Water supply	Binary (0 or 1)
(Land cover)		
LC_BCH	Beach	Binary (0 or 1)
LC_CRL	Coral Reefs and atolls	Binary (0 or 1)
LC_EST	Estuary	Binary (0 or 1)
LC_FWT	Nearshore freshwater wetland	Binary (0 or 1)
LC_ILD	Nearshore Islands	Binary (0 or 1)
LC_50M	Nearshore Ocean--50m depth or 100km offshore	Binary (0 or 1)
LC_OPS	Open ocean	Binary (0 or 1)
LC_SWT	Saltwater wetland, marsh or pond	Binary (0 or 1)
LC_GRS	Seagrass beds or kelp forests	Binary (0 or 1)
LC_SMI*	Semi-enclosed seas	Binary (0 or 1)
(Geopolitical region)		
SP_OCE	Oceania	Binary (0 or 1)
SP_NA	North America	Binary (0 or 1)

SP_SA	South America	Binary (0 or 1)
SP_EU	Europe	Binary (0 or 1)
SP_AS*	Asia	Binary (0 or 1)
MARQUEE_STATUS	Whether a national park, RAMSAR site etc.	Binary (0 or 1)
URBAN	Whether an urban area	Binary (0 or 1)
STUD_AREA	Area of the study site	Continuous
--Situation of valuation		
(Type of change)		
MG_OTHER	Change in other areas	Binary (0 or 1)
MG_WATER	Change in water resource management	Binary (0 or 1)
MG_FISH	Change in fish population etc.	Binary (0 or 1)
MG_WILD	Change in wildlife management	Binary (0 or 1)
MG_INFRA*	Change in infrastructure	Binary (0 or 1)
(Degree of change)		
CHG_0	No change	Binary (0 or 1)
CHG_1	Improvement step 1	Binary (0 or 1)
CHG_2	Improvement step 2	Binary (0 or 1)
CHG_-1*	Undesirable change	Binary (0 or 1)
--Subject of valuation		
INCOME	Income	Continuous
POP_DEN	Population density	Discrete
Methodology consistency		
(Elicitation method)		
ELI_DM	Dichotomous choice	Binary (0 or 1)
ELI_OD	Open end	Binary (0 or 1)
ELI_ITR	Iterative bidding	Binary (0 or 1)
ELI_PCD	Payment card	Binary (0 or 1)
ELI_CB	Contingent behavior or combined	Binary (0 or 1)

	CV& Revealed Preference (RP)	
ELI_CK*	Contingent ranking	Binary (0 or 1)
INTERVIEW	Whether phone or impersonal interview was applied	Binary (0 or 1)
(Payment vehicle)		
VHC_MKT	Market based payment e.g. water bill	Binary (0 or 1)
VHC_TAX	Tax	Binary (0 or 1)
VHC_DNT*	Donation	Binary (0 or 1)
NONUSERS_ONLY	Whether the sample population only including nonusers	Binary (0 or 1)
LUMPSUM	Whether it is a lump sum payment	Binary (0 or 1)
SUBSAMPLE	Whether outliers was excluded	Binary (0 or 1)
MEDIAN	Whether it is a median value	Binary (0 or 1)
STUBSTITUTION	Whether substitution included	Binary (0 or 1)
Quality of the study		
PRIMARY_DATA_ONLY	Whether external data used in calculating per unit value	Binary (0 or 1)
SURVEY_YEAR	Year of the study	Discrete

* These variables were omitted from all regressions in order to avoid collinearity due to dummy variables summing to unity. Therefore, all effects are measured relative to a base case with these characteristics.

Table 2: Mean, median and Standard Deviation (SD) of WTP estimates by service, land cover, geopolitical region, and elicitation method (Unit: 2006 US $ household^{-1} year^{-1})

Variable (number of observations)	Mean WTP	Median	SD
Ecosystem services			
Aesthetic (20)	3268	600	6024
Disturbance control (1)	1.5	1.5	0
Food (1)	0.3	0.3	0
Habitat (18)	51	48	28
Recreation (50)	426	121	932
Spiritual (9)	39	32	36
Water quality (21)	192	112	207
Land cover			
Beach (25)	38	19	33
Coral Reefs and atolls (9)	812	766	574
Estuary (16)	1222	195	3964
Nearshore freshwater wetland (6)	152	110	185
Nearshore Islands (4)	37	35	9
Nearshore Ocean--50m depth or 100km offshore (28)	522	137	1169
Open ocean (6)	310	83	392
Saltwater wetland, marsh or pond (21)	2189	127	5201
Seagrass beds or kelp forests (3)	179	24	279
Semi-enclosed seas (2)	53	53	6
Geopolitical region			
Oceania (4)	105	89	76
North America (94)	949	115	3060
South America (1)	89	89	0
Europe (9)	48	48	28
Asia (12)	151	40	277

Elicitation method			
Dichotomous choice (45)	349	109	935
Open end (23)	88	32	150
Iterative bidding (11)	37	19	38
Payment card (13)	60	48	41
Contingent behavior (16)	702	758	508
Contingent ranking (12)	5149	806	7273

Table 3: Comparison of different models

	Linear		Box-Cox		Log-log		Stepwise log-log	
	Coeff	**p**	**Coeff**	**p**	**Coeffi**	**p**	**Coeff**	**p**
Income	-0.009	0.46	0.35	0.00	0.37	0.43	0.42	0.06
Density	-1.22	0.00	0.11	0.00	0.11	0.37	0.09	0.17
Area	0.004	0.05	0.18	0.00	0.19	0.00	0.17	0.00
Survey year	31.46	0.77	-0.05	0.00	-0.052	0.36	-0.05	0.04
Constant	149056	0.01	4.12	0.00	3.81	0.53	3.94	0.19
Lambda	NA	NA	0.004	0.19	NA	NA	NA	NA
Residual Statistics								
Skewness	0.32	0.16	-0.64	0.00	-0.66	00.0	-0.64	00.0
Kurtosis	1.63	0.00	2.78	0.00	2.88	0.00	2.44	0.00
Jarque-Bera	15.21	0.00	46.86	0.00	50.2	0.00	37.8	00.0
Breusch Pagan heteroskedasticity Test								
Chi-Squared	75.09	0.006	57.19	0.15	57.2	0.15	30.88	0.19

Table 4: Meta-regression result of the step-wise log-log model (N=120, df = 94, R^2 = 0.79)

	Variable	Coeff	Significance Level
1	LNINCOME	0.42	0.060
2	LNDENSITY	0.09	0.165
3	LNAREA	0.17	0.000
4	Constant	3.94	0.189
5	SURVEY_YEAR	-0.05	0.036
6	ES_FOOD	-5.44	0.000
7	ES_SPR	-0.76	0.078
8	BUNDLED_SERVICES	-0.36	0.112
9	SP_OCE	-1.22	0.001
10	SP_SA	2.71	0.000
11	SP_EU	0.85	0.024
12	LC_BCH	-1.48	0.000
13	LC_EST	-0.45	0.092
14	LC_OPS	-0.60	0.027
15	CHG_0	-0.98	0.010
16	CHG_1	-1.24	0.001
17	CHG_2	-0.93	0.024
18	ELI_DM	-2.30	0.000
19	ELI_ODD	-2.50	0.000
20	ELI_ITR	-3.00	0.000
21	ELI_PCD	-4.21	0.000
22	ELI_CVBR	-1.82	0.000
23	INTERVIEW	-0.43	0.049
24	VHC_TAX	0.27	0.107
25	LUMPSUM_PAYMENT	1.37	0.000
26	SUBSAMPLE	-0.42	0.048

Figure 1: Actual and predicted WTP values

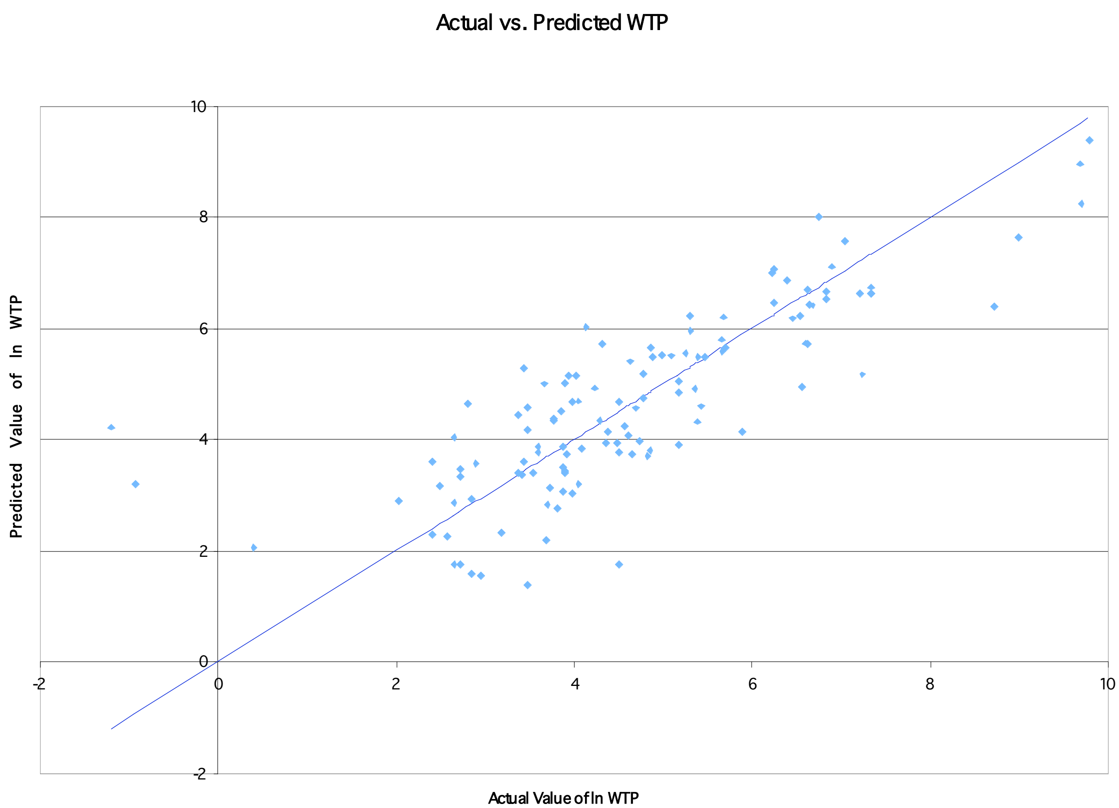
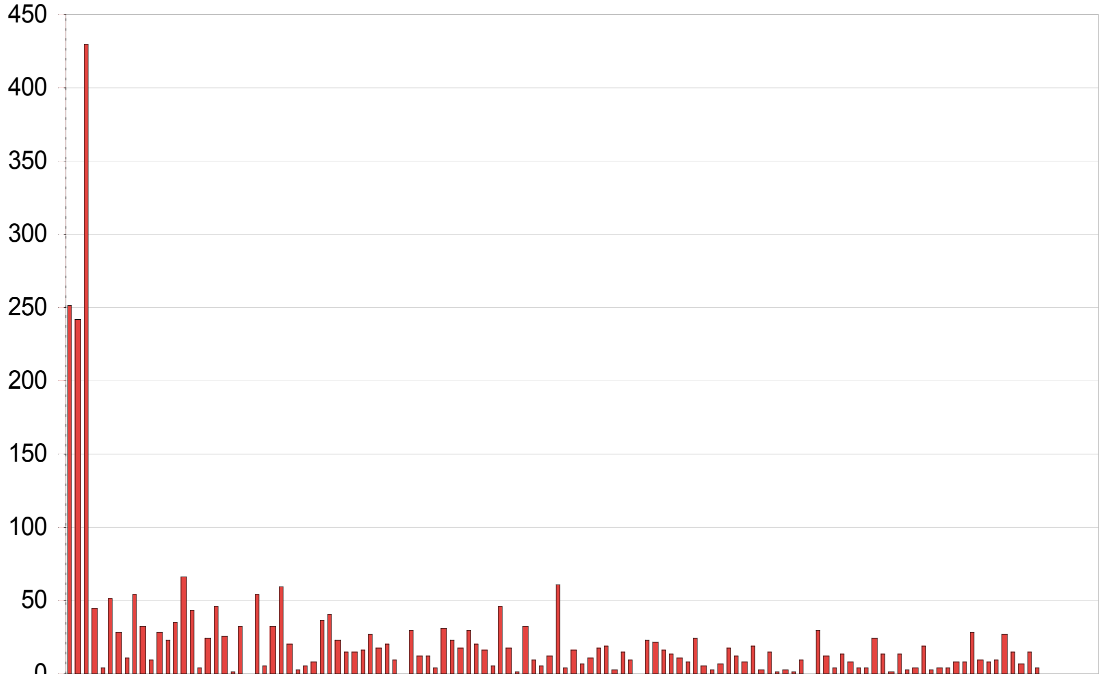


Figure 2: Transferred error associated with each observation ranked in an ascending order



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Ecological and Economic Roles of Biodiversity in Agroecosystems*

M. Ceroni, S. Liu, and R. Costanza

As ecosystems become less diverse as a consequence of land conversion and intensification, there is a shared concern over the functioning of these systems and their ability to provide a continuous flow of services to human societies (Ehrlich and Wilson 1991). The ecological consequences of biodiversity loss on ecosystem functioning have been investigated for more than a decade, but only recently has interest developed around the consequences of agricultural biodiversity loss on the functions of agroecosystems. Agricultural intensification has led to a widespread decline in agricultural biodiversity measured across many different levels, from a reduction in the number of crop and livestock varieties, to decreasing soil community diversity, to the local extinction of a number of natural enemy species.

Each time species go locally extinct, energy, and nutrient pathways are lost with consequent alteration of ecosystem efficiency and of the ability of communities to respond to environmental fluctuations. Monocultural agroecosystems typically display low resilience to perturbations such as drought, flooding, pest outbreaks, and invasive

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* This paper was published as chapter 18 in Jarvis, D. I., C. Padoch, and D. Cooper (eds). *Managing Biodiversity in Agroecosystems*. 2005. New York: Columbia University Press.

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species and to uncertainties related to market fluctuations. Large inputs of energy are then needed in the form of fertilizers, pesticides, herbicides, and irrigation.

Multifunctional and sustainable agriculture, where production is achieved with respect for ecosystem functions and processes and with reduced impacts to other systems, is expected to produce a whole array of ecosystem services besides edible and fiber biomass production, such as soil erosion control, carbon sequestration, nutrient cycling, wildlife refugia, and sources of spiritual and cultural enjoyment.

Ecosystem functioning refers to the rates and magnitudes of ecosystem processes, such as primary production, decomposition, and nutrient cycling. Ecosystem services are the functions that directly or indirectly affect human welfare. Whereas well-established measures of ecosystem functioning exist, such as mineralization rates and organic matter production, it is difficult to translate what ecologists measure into ecosystem services. Because ecosystem services represent anthropocentric properties of the ecosystems, the notion of value is inherently part of their definition. For this reason ecosystem services often are measured in economic terms rather than in ecological terms of energy and material flux (see Costanza et al. 1997). Although local and global economies depend heavily on ecosystem services, these have been traditionally ignored by commercial markets and therefore have been given little weight in policy decisions.

This is well exemplified by the study conducted by Costanza and colleagues (1997) on the economic value of ecosystem services at the global level. The study estimated the total economic value of ecosystem services for the globe based on economic valuations of ecosystem services for each of 16 biomes (communities of plants and animals that are well adapted to different climatic regions of the earth, such as deserts, grasslands, or temperate forests).

The authors found that estimates of global economic activities, such as the yearly global gross national product, failed to account for the substantial economic contribution of ecosystem services from the different world's biomes. Whereas global gross national product was estimated to be around US$18 trillion per year, the economic value of ecosystem services ranged between US$16 and 54 trillion per year, with an average of US$33 trillion per year (in 1994 U.S. dollars). It is not well understood from this study to what extent different agricultural ecosystems contributed to the total value of ecosystem services. Croplands, with a total value of US$128 billion per year (0.38% of total estimated value), seem to contribute little to the global flow of ecosystem services beyond food production (table 18.1).**[[Table 18.1 about here.]]** However, this result is mainly a consequence of the limited information available on ecosystem services in food production systems and of the assumption that croplands do not provide habitat for wildlife, nor do they represent a valuable source for recreation. When grass and rangeland systems are included, most of which are assumed to be subject to various levels of grazing for farming purposes, the total value of annual ecosystem services from agricultural lands jumps to US$1.03 trillion (3.1% of total estimated value). Croplands and grass and rangelands together contribute mainly to food production (US$336 billion), followed by biological control (US$121 billion) and pollination services (US$117 billion). The main services contributed by the grass and rangeland component of agricultural lands are waste treatment (US$339 billion) and erosion control (US$113 billion). Given the large scale of this study and the broad categories used to identify the main biomes, these figures do not capture the role of the different agricultural land uses (e.g., shrimp farming, aquaculture, flooded fields, and agroforestry), inevitably underestimating the contribution of the world's agroecosystems.

No matter what the final figures of the total contribution of agroecosystems to human welfare would be, agricultural biodiversity is what supports the ecosystem services that our societies depend on. Yet estimating the specific economic contributions of agricultural biodiversity and biodiversity in general to the value of ecosystem services is a formidable challenge (see Turner et al. 2003 and Smale 2005).

For the sake of economic valuation of biodiversity, a distinction can be made between biological resources and biological diversity (OECD 2002). Biological *resources* are elements of ecosystems, such as genes or species, which are of direct importance to human economies. Biological *diversity* is considered to be of value to human societies as the source of the variety of species' ecological interactions, physiological tolerances, structural arrangements in space, and genetic structures that in the end determine ecosystem functioning.

The importance of economic valuation of biodiversity is recognized by the Convention on Biological Diversity (CBD). CBD's Conference of the Parties Decision IV/10 recognizes that "economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated incentive measures."

Most studies on biodiversity valuation have assessed the direct value of *biological resources* (i.e., the value that is more readily captured by commercial markets), focusing in particular on plant or crop and animal genetic resources or the direct use of plant species for medicinal or ornamental use (for the direct value genetic resources in crop improvement, see reviews in Alston et al. 1998; Evenson and Gollin 2003). The nonmarket values of genetic resources have been assessed in a very few cases, including livestock genetic resources (Drucker, chapter 17) and, most recently, components of agricultural biodiversity in home gardens (Birol 2004; Birol et al. 2004). Two collections of studies about valuing crop genetic resources conserved in banks (Koo et al. 2004) and the biological diversity of crop plants on farms (Smale 2005), both using primary data, have been published recently. Although they are based on detailed field research, these studies advance methods for valuing only a few components, or entry points, of biological diversity.

Almost no information exists on the economic value of most components of biological diversity to human societies and, particularly, their indirect value. For example, the diversity in species or functional groups in an ecological community is of value to our society to the extent that it matters to the provision of the services we benefit from, such as nutrient cycling, biomass production, and stability of biomass production. But proving that community diversity does actually matter is extremely difficult, and even more difficult is to identify general ecological rules that can fit the broad purposes of economic valuation. In this chapter we report results from empirical ecological studies that measured the relationship between diversity and ecosystem functions (mostly in agricultural systems), under the assumption that measures of ecosystem functions provide a useful indication of the direction and intensity of the flow of ecosystem services without necessarily translating directly into ecosystem services. We focus primarily on the role of agricultural biological diversity (instead of biological resources). Besides providing evidence from empirical ecological studies, each section briefly addresses how ecological knowledge of agrobiodiversity can be applied to inform economic valuation. Valuation methods for biodiversity and ecosystem services have been extensively reviewed recently (Wilson 1988; Orians et al. 1990; Drucker et al. 2001; Nunes and van den Bergh 2001), so methodological considerations are not part of our discussion. We begin the chapter with an overview of the main concepts and findings from a decade of biodiversity and ecosystem functioning literature. We then discuss how agrobiodiversity relates to stability and resilience in agricultural systems. The role of habitat heterogeneity to support wild species is then examined, followed by a section on agrobiodiversity at the landscape scale. We conclude with observations on research needs in assessing the relationship between agrobiodiversity and ecosystem services and implications for agrobiodiversity economic valuation studies.

Diversity of Producers and Biomass Production

Over the last decade, the most influential empirical research on the links between biodiversity and ecosystem function has been the series of experiments manipulating plant species diversity and functional group richness in grasslands (e.g., Naeem et al. 1994; Tilman et al. 1996, 2002; Hector et al. 1999) and in aquatic microbial microcosms (reviewed by Petchey et al. 2002).

Because recent publications cover biodiversity functioning research extensively (Chapin et al. 2000; Loreau et al. 2001, 2002; Kinzig et al. 2002; see also chapters 9 and 10), we only briefly review the central issues.

Empirical and theoretical studies in many cases have confirmed associations between biodiversity and ecosystem functioning, but many relationships, from insignificant to significant, from positive to negative, have been identified depending on the scale of the investigation (Naeem 2001). Many factors, such as site fertility, disturbance, habitat size, climate (Wardle et al. 1997), the presence or absence of trophic groups (Mulder et al. 1999; Naeem et al. 2000), and the functional composition of species (Hooper and Vitousek 1997; Tilman et al. 1997a), can determine the relationship between biodiversity and ecosystem function.

Several studies found significant positive correlations between species richness and plant biomass accumulation (reviewed by Schmid et al. 2002). The mechanisms behind these correlations were long debated around two main hypotheses, although alternative explanations also have been discussed (reviewed by Eviner and Chapin 2003). Aarssen (1997), Huston (1997), and Tilman et al. (1997b) suggested that the often-observed increase in primary productivity in more diverse plots may have reflected a sampling effect. A community with a higher number of species inherently has a higher probability of including species with superior traits. Another explanation of diversity effects on ecosystem functioning is niche complementarity (Naeem et al. 1995; Tilman et al. 1997a). Higher species diversity in a community increases the range of ecological traits—and consequently the variety of niches available—leading to a more efficient resource use in a variable environment. Recently, the debate appears to have been reconciled (Loreau et al. 2002; Naeem 2002). Niche complementarity and sampling effects seem to play different roles in different phases of the experimental manipulations: Initially, a rapid growth response that seems compatible with the sampling mechanism is observed, with the best diversity plots reaching a productivity almost equal to that of the best monocultures. After two or more years, a longer-term response shows the best diversity plots producing higher yields than the best monocultures, a pattern that can be explained by interspecific competition resulting from niche differentiation (Pacala and Tilman 2002).

One general conclusion seemed to emerge among the contrasting results and interpretations that a decade of diversity functioning research has generated: The species' role in the functioning of these experimental communities can vary widely. Some species might be indispensable in maintaining the functioning of an ecosystem, as in the case of keystone species (Paine 1966) or ecosystem engineers (Jones et al. 1994; Wright et al. 2002). Some other species may even appear redundant in their ecological functions and may be easily replaced by other species with no appreciable consequences for ecosystem functioning, should they go locally extinct (Walker 1992; Gitay et al. 1996; Naeem 1998).

As discussed also in chapter 10, one of the limitations of biodiversity function studies is that they have been performed in small, controlled patches that are far from mimicking the conditions of natural or even managed ecosystems. For example, it is hard to extrapolate the implications of this type of research for agricultural systems, where the number of crop species used typically is low and rotation cycles govern the temporal dynamics of the system.

Very few experiments have manipulated species richness in agricultural systems to assess the effects on biomass production. Results from a study on hay fields in southern Britain show that restoration of species richness in fields that were previously impoverished in species had a positive effect on hay production. Bullock and colleagues (2001) reported a 60% yield increase in species-rich treatments in hay meadow restoration experiments at seven sites across southern Britain. At each site two seed mixes (species poor, with 6 ± 17 species, and species rich, with 25 ± 41 species) were applied in a randomized block experiment. Hay yield was higher in the species-rich treatment from the second year onward, by up to 60% (figure 18.1).**[[Figure 18.1 about here.]]** Comparing the two treatments in all sites, there was a simple linear relationship between the difference in species number and the amount of increase in hay production. Fodder quality was the same in both treatments. This suggests that farmers can maximize high-quality herbage production in resown grasslands by maximizing biodiversity. The results of this study are particularly remarkable if we think that there is a common misconception among farmers that every effort to increase biodiversity results in lower food production.

The only apparent shortcoming in this study was the higher cost of the high-diversity seed mix; a higher increase in yield would be needed to offset these additional costs. The ecological mechanisms behind the observed patterns seem to be a result of species number differences between treatment and control plots, but the authors warn that because species number and composition were not varied independently (as done by Hector et al. 1999), compositional differences also might have contributed to yield differences.

Economic Considerations

In this study case, the economic contribution of species richness to hay production is straightforward to assess as the difference between production outcomes under the two different richness treatments. Valuations of this kind could be used to develop incentives to farmers to promote higher plant diversity in hayfield systems.

In most cases, though, assessing the economic contribution of crop species richness to other ecosystem services such as nitrogen cycling or CO_2 regulation is not as straightforward. In the best-case scenario, even assuming that the ecological causalities between agrobiodiversity and ecosystem functions have been clearly identified, economic assessments would rarely reach a validity that goes beyond the scale of the studied site.

Attempts are being made to assess the specific ecological and economic contributions of species richness to net primary productivity and nutrient cycling in natural and seminatural environments at a regional scale based on multiple regression models (Costanza et al. unpublished).

Diversity of Consumers and Decomposers

Most studies have focused on the role of the diversity of primary producers in providing fundamental ecosystem services. However, very little is known about the factors influencing ecosystem services provided by higher trophic levels in natural food webs. A recent study of 19 plant–herbivore–parasitoid food webs (Montoya et al. 2003) showed that differences in food web structure and the richness of herbivores influence parasitism rates on hosts, promoting the service supplied by natural enemies. One main result of this study was that parasitoids function better in simple food webs than in complex ones, indicating that species richness per se might not be a key factor in the provision of higher-level ecosystem services when more complex, multitrophic communities are investigated.

As Brown et al. (chapter 9) noted, most evidence suggests that in soils there is no predictable relationship between species diversity and specific soil functions, making it difficult to foretell the consequences of decreased soil species richness (Mikola and Setälä 1998). In many cases, soil ecosystem function seems to be controlled by individual traits of dominant species and by the complexity of biotic interactions that occur between components of soil food webs (Cragg and Bardgett 2001).

Higher functional diversity in microbial communities has been associated with higher efficiency in resource use. For example, a 21-year study comparing biodynamic, organic, and conventional farming systems in central Europe (Mäder et al. 2002) shows that more diverse microbial communities, typical of organically managed soils, transformed carbon from organic debris into biomass at lower energy costs.

Economic Considerations

In systems where the role of one individual species determines the rate of a given set of ecological processes and the flux of a given ecosystem service, that species could be valued independently. However, this is rarely the case. Complex ecological interactions normally make it difficult to disentangle the role of particular species and the effect of diversity per se in supporting certain ecosystem functions. For these reasons, ecological economists have tended to value biodiversity indirectly by valuing the services biodiversity supports. For example, Walker and Young (1986) estimate that soil erosion was responsible for revenue loss from agriculture in the Palouse region, northern Idaho and western Washington, in the range of US$10 to US$15 per hectare. This estimate is an aggregated indicator of the ecological functions responsible for erosion control in agroecosystems of that particular region.

Diversity and Resilience in Agroecosystems

Most studies on biodiversity and ecosystem functioning have been conducted in stable conditions. Agroecosystems typically are subject to cyclical perturbations of variable intensity as a consequence of agricultural practices and to unpredictable events such as pest outbreaks and drought. However, the relationship between diversity and ecosystem function might change in a fluctuating environment (see chapters 13 and 14).

There is a general agreement that a major role of biodiversity in relation to ecosystem services is insurance against environmental change (e.g., Holling et al. 1995; Perrings 1995). A higher number of functionally similar species ensures that when environmental conditions have turned against the dominant species, other species can readily substitute for their functions, thereby maintaining the stability of the ecosystem (Yachi and Loreau 1999) and enhancing ecosystem reliability (i.e., the probability that a system will provide a consistent level of performance over a given unit of time) (Naeem and Li 1997).

For example, diversity of pollinators is essential to food production systems, not only because pollen limitation to seed and fruit set is widespread (Burd 1994) but, most importantly, in the face of the ongoing trends of pollinator disruptions (Nabhan and Buchmann 1997; Kremen and Ricketts 2000; Cane and Tepedino 2001; see also chapter 8). Kremen et al. (2002) found that a diversity of pollinators was a determinant for sustaining pollination services in conventional (versus organic) farms in California because of annual variation in composition of the pollinator community.

Redundancy in soil microbial communities seems to be very common and crucial in maintaining soil resilience to perturbations (see chapter 9). For example, experimental reductions of soil biodiversity through fumigation techniques show that soils with the highest biodiversity are more resistant to stress than soils with impaired biodiversity (Griffiths et al. 2000).

Studies conducted in extreme regions of the world, such as the Dry Valley in Antarctica, where soil communities are much less diverse, provide unique experimental sites to address the role of food web complexity in soil function. Nematode communities in this region, comprising three species at the most, typically lack redundancy and are particularly sensitive to environmental change (Freckman and Virginia 1997).

Agrobiodiversity at the genetic level also provides an insurance value in the face of changing environmental conditions. Chapters 2 through 6 describe empirical evidence of how in food production systems, genetic diversity ensures adaptability and evolution by providing the raw material for desirable genetic traits in crops and livestock. In chapter 15, Johns demonstrates how agricultural diversity and the knowledge imbedded in its management are essential for dietary diversity and human health.

Ecosystems that are capable of absorbing a higher degree of perturbation before their functioning is significantly altered (i.e., are more ecologically resilient, *sensu* Holling 1973) can provide ecosystem services more consistently. Planting of varietal mixtures with differing levels of pest resistance has proved to be a successful strategy to fight fungal pathogens (see also chapters 11 and 12 and Zhu et al. 2002).

Resilience in industrial monocultures is achieved through use of external inputs such as chemical fertilizers, pesticides, and fossil fuels. As noted in chapters 12, 13, 16, and 17, in less intensive systems agricultural biodiversity may provide a buffer to unpredictable environmental and market fluctuations. Several scientists have urged recognition of the indissoluble link between ecological and sociological resilience in managed systems (Scoones 1999; Folke et al. 2003; Milestad and Hadatsch 2003). In fact, systems may be ecologically resilient but socially vulnerable or socially resilient but environmentally degrading (Folke et al. 2003). Agricultural systems can then be thought of as social-ecological systems that behave as complex adaptive systems, in which the managers are integral components of the system (Conway 1987). In chapter 13 the term *agrodiversity* is used to interrelate agrobiodiversity, management diversity, and biophysical diversity into organizational diversity. To be resilient to natural and market fluctuations, agroecosystems should withstand disturbance, be able to reorganize after disturbance, and have the ability to learn and adapt in the face of change (Walker et al. 2002). Exponents of the Resilience Alliance argue that resilience is something that can and should be managed to “prevent the system from moving to undesired system configurations in the face of external stresses and disturbances” and to “nurture and preserve the elements that enable the system to renew and reorganize itself following a massive change" (Walker et al. 2002). Both ecological components and human capabilities can play an important role in resilience management. For example, the insurance value of agricultural biodiversity has a recognized role in protecting ecosystem resilience (Heywood 1995). Furthermore, agricultural systems with high levels of social and human assets are more flexible and more capable of incorporating innovations in the face of uncertainty (Pretty and Ward 2001).

Economic Considerations

Identifying and measuring the insurance value of biodiversity is a far from trivial exercise. For example, what premium would be paid to preserve resilience in a given system? One option would be to consider the cost of maintaining a nonresilient system. In agroecosystems this premium would be equivalent to the entire costs of maintaining intensive agricultural practices through the use of external inputs, including costs of pesticides and chemical fertilizers. As noted earlier in this chapter and in chapter 8, diversity of pollinators is needed to maintain the resilience of production systems in the face of declining pollinators. Southwick and Southwick (1992) calculated for each of 62 U.S. crops the extent to which wild pollinators could replace honeybee functions, should they decline to the degree predicted by their model. In the absence of compensation from wild pollinators, alfalfa yield losses were estimated to be 70% of total production, equivalent to US$315 million a year.

Maintaining or enhancing the insurance function of species and genetic diversity might come with a cost to other functions that are relevant to human welfare, such as food and fiber production. For example, Heisey et al. (1997) assessed the yield losses associated with switching to a more genetically diverse portfolio of wheat varieties in Pakistan at tens of millions of U.S. dollars per year. Widawsky and Rozelle (1998), Di Falco and Perrings (2003), Meng et al. (2003), and Smale et al. (1998) found both positive and negative associations between crop variety diversity, crop productivity, and yield variability, depending on the cropping system context. Whereas the insurance value of genetic diversity in food production systems has been assessed at least in some cases (e.g., see the studies assessing costs of conservation programs for genetic resources reviewed by Drucker et al. 2001), there are no studies addressing the insurance value of a diversified portfolio of functions and phenotypic traits provided by crop species, soil organisms, or natural enemies. The difficulties in determining the insurance value are related to the intangible nature of this service and the inability to account for future benefits adequately. In addition, the outcomes of a valuation study might vary according to the perceived level of collapse threat.

Agricultural Habitats and Landscape Diversity

Various studies show that agricultural landscape diversity can reduce yield losses to pests by affecting populations of both herbivorous insects and natural enemies (see Andow 1991 for a review). For example, healthier populations of predator carabid beetles can be found in more heterogeneous farm systems (where heterogeneity is measured as perimeter-to-area ratio) and in systems with higher crop species diversity (Ostman et al. 2001).

The composition and spatial arrangement of perennial and annual crops in the agricultural landscape can also be crucial for the long-term predator–population dynamics (Bommarco 1998; Thies and Tscharntke 1999).

In other cases polycultures do not seem to provide any advantage to natural enemy populations when compared with monocultures (Tonhasca and Stinner 1991).

Inconsistent results in experiments that have manipulated landscape structure and vegetational diversity might reflect the variation related to the different spatial scale of the experimental vegetation plots. A comprehensive meta-analysis of the literature results in this field over a period of 18 years shows that in experiments performed in small plots, spatial heterogeneity tends to have a large negative effect on herbivores, intermediate-sized plots show an intermediate effect, and the largest plots exhibit a negligible effect (Bommarco and Banks 2003).

Finding general patterns in the relationship between landscape diversity and species diversity becomes even more complicated when diversity across multiple taxa is investigated (Tews et al. 2004 and references therein; see also chapters 13 and 14). This relationship specifically depends on at least three factors: the species groups studied, the measurement of landscape diversity, and the temporal and spatial scales.

More diverse agricultural landscapes provide important habitats not only for natural enemies but also for pollinators, enhancing the provision of pollination services (see also chapter 8). A study on the effects of agricultural landscape structure on bees found that species richness and abundance of solitary wild bees were positively correlated with the percentage of seminatural habitats, an indicator of landscape diversity (Steffan-Dewenter et al. 2002). The correlation depended on spatial scale and species group. For example, whereas solitary wild bees responded to landscape complexity at the small scales, honeybees were correlated with landscape structural characteristics only at large scales. In other cases, the availability of suitable foraging habitats matters more than landscape heterogeneity in determining the species richness of pollinators (Steffan-Dewenter 2003).

Bird and mammal species richness also can be enhanced by agricultural landscape diversity. A recent review (Benton et al. 2003) provides ample evidence that habitat heterogeneity matters to farmland biodiversity from the individual field to the whole landscape. For example, seed-eating birds seemed to occur in higher numbers in pastoral areas containing small patches of arable land than in pure grassland landscapes (Robinson et al. 2001). Some bird species specifically depend on the open habitats provided by farming systems in Africa (Söderström et al. 2003), as in Europe (Pain and Pienkowski 1997) and Central America (Daily et al. 2001).

Agroforestry patches can harbor a number of wild species similar to or higher than that of original forest patches. For example, Ricketts et al. (2001) found no significant difference in the abundance and richness of moth species between forest and agricultural fragments composed of coffee monocultures, shade-grown coffee, pasture, and mixed farms. Polycultural coffee plantations designed to mimic natural systems in various cases show species richness equal to or greater than that of adjacent natural forest patches (figure 18.2) (Perfecto et al. 1997; Daily et al. 2003).**[[Figure 18.2 about here.]]** A decline in species diversity in agroforests can be observed with increasing distance from the forest patches (Ricketts et al. 2001; Armbrecht and Perfecto 2003), although this result is not consistent across studies (e.g., Daily et al. 2003). In Central and South America, shaded coffee plantations that include leguminous, fruit, fuelwood, and fodder trees are reported to contain more than 100 plant species per field and support up to 180 bird species (Michon and de Foresta 1990; Altieri 1991; Thrupp 1997).

Noncultivated areas (e.g., riparian buffers, windbreaks, or border plantings), improved fallows, and woody vegetation play an important role in maintaining biodiversity of weeds, insects, arthropods, and birds (Benton et al. 2003 and references therein; McNeely and Scherr 2003). Hedgerows and woody vegetation, while providing habitats for wild biodiversity, may enhance other ecosystem services such as soil stabilization, soil erosion control, and carbon sequestration.

Economic Considerations

Once a strong link between agricultural habitat diversity and wild species diversity has been documented, the value of agrobiodiversity to wildlife habitat protection can be assessed through the expenditures associated with the enjoyment of a biologically richer environment. Alternatively, assessments can include the costs of protecting the diversity of habitats that agrobiodiversity provides. For example, citizens in the Netherlands were willing to pay between 16 and 45 guilders per household per year (corresponding to $10.80 and $30.35 in 2003 U.S. dollars) to fund management practices that would enhance wildlife habitat in the Dutch meadow region (cited in Nunes and van den Bergh 2001).

Recreational and Cultural Roles of Agricultural Biodiversity

A variety of different agricultural land uses can promote scenic beauty, with positive effects on the economy of local communities. For example, it is known that aesthetic properties are associated with heterogeneity in the landscape (Stein et al. 1999). Entire communities in the Tuscany region, in Italy, benefit from a rural tourism economy that is based on the diversity of agricultural patches ranging from vineyards, wheat fields, pasture lands, and orchards to olive tree cultivations. Similarly, the Montado, in the Alentejo region of southern Portugal, is a highly diverse agricultural landscape. Cork and helm oaks are grown in varying densities, combined with a rotation of crops, fallows, and pastures, providing natural, scenic, and recreational value (Pinto-Correia 2000). Another example of an agriculturally rich region is the Pinar del Rio province in Cuba, where a healthy agrotourism industry relies on different natural attractions interspersed in a mosaic of agricultural lands, integrating tobacco fields, sugarcane cultivations, and fruit trees (Honey 1999). Various European countries and states in the United States have policies to preserve the traditional character of agricultural landscapes. For example, Switzerland subsidizes farmers in mountain areas to maintain a mix of agricultural and natural land covers because of the recreational value of these heterogeneous systems (McNeely and Sherr 2003). Conservation organizations such as the Land Trust in the United States often use the purchase of development rights as a way to maintain the rural, multiuse character of agricultural landscapes, which is perceived as a source of recreational activities and cultural enjoyment.

Agricultural biodiversity is a crucial source of nonmaterial well-being that derives from nutrition traditions, dietary diversity, and longstanding knowledge (chapter 15). Plant and animal diversity in small-scale farming often can serve the purpose of personal enjoyment or the fulfillment of family or clan tradition or may meet spiritual needs. For example, the variety of domesticated plants and livestock breeds in various regions of the world have provided raw materials for artistic expression in textiles and other crafts for centuries. As another example, home gardens are cultivated not only for food production but also with ornamental and aesthetic values in mind (Kumar and Nair 2004).

Economic Considerations

A comprehensive assessment of the value of landscape agricultural diversity for recreational purposes has not been conducted. However, data sources abound for recreational expenditures in regions that comprise a variety of agricultural land uses (e.g., Fleischer and Tsur 2000). Alternatively, the value of agricultural landscape heterogeneity might be assessed by surveys to estimate the economic value that visitors would place on the maintenance of the landscape. For example, Drake (1992) found that Swedish citizens were willing to pay US$130/ha each year to preserve agricultural land against conversion into forest, a value that was higher than the return from agricultural production in most regions of Sweden.

Whereas ecologists have identified measures of ecosystem functions (such as biomass for primary productivity or mineralization rates for nitrogen cycling), there are no corresponding quantities that can be used as measures of social function related to agricultural diversity. In many rural societies the cultural value of certain plant species resides beyond any notion of monetary measure. It may be argued that intrinsic values for these plant uses cannot be measured. These are cases in which monetary valuations of biodiversity services may be inappropriate. Alternative valuation methods that are relevant to policy and decision making must be developed for these kinds of contributions. An initial step in this direction is represented by a recent study assessing the historical and cultural value of livestock diversity in Italy (Gandini and Villa 2003). The authors qualitatively evaluated nine local cattle breeds based on their value to folklore, gastronomy, handicrafts, and the maintenance of local traditions.

Conclusion

The services that agricultural biodiversity provides are critical to the functioning of food support systems. They contribute to human welfare, both directly and indirectly, and therefore represent part of the total economic value of the planet.

There is a general agreement that the management of agricultural biodiversity can provide ways to increase food production while beneficially affecting other ecosystem services. Multifunctional and sustainable agriculture are expected to produce higher flows of ecosystem services, but the extent of these contributions and their economic value has yet to be quantified.

The positive results from studies of multifunctional agricultural systems often are overlooked because these results normally are achieved at a small scale and are difficult to document. Nonetheless, small-scale farming is the predominant form of farming in many regions of the world and is projected to remain so in marginal areas where little investment in new agricultural technologies is expected to occur (Wood et al. 2000). Identifying alternative experimental models may be crucial if more conclusive understanding of the relationship between agrobiodiversity and ecosystem functions and services is to be achieved. For example, it is well understood that large-scale experiments in agriculture (involving hundred of small farmers) might take place only as a result of a strong political will and where economic benefits for the farmers involved are clearly prospected, as in the case of the use of mixed rice varieties in the Yunnan province of China (Zhu and colleagues, chapter 12).

Often, however, the benefits to small farmers of experimenting or adopting new practices to maintain agrobiodiversity on their land might not be immediately available or apparent. This is especially the case for the values of agrobiodiversity that are not directly traceable in the marketplace. These include the insurance value against risk and uncertainty, the value of supporting relevant ecosystem services, and the cultural and aesthetic functions. A full assessment of these values (that includes monetary as well as ecological evaluations) is key to encouraging decision makers to invest in programs for the active protection and maintenance of agrobiodiversity. In particular, economic valuations of nonmarket benefits of agrobiodiversity can be used to identify incentives for farmers to adopt innovative cultivation methods that might be beneficial for agrobiodiversity but might not be economically viable.

In general, current valuation methods must be supported by a better understanding of the relationships between agrobiodiversity and ecosystem functions and by the identification of the functions that are irreplaceable.

Recent developments in the field of ecosystem service valuation show geographic information system–based spatial representation of valuation data as a valuable visualization tool to facilitate management planning and to identify target areas for conservation. For example, in a study commissioned by the Audubon society in Massachusetts, researchers M. Wilson and A. Troy were able to visualize nonmarket values of ecosystem services at the watershed level (Breunig 2003).

So far, valuation studies conducted at a regional scale do not differentiate between the various agricultural land uses, making it difficult to assess the economic value of ecosystem services provided by agricultural ecosystems at the larger scale.

Whenever used to inform and redesign policy, economic valuation studies of agrobiodiversity should be regarded as indicative estimates, recognizing the uncertainties about the actual contributions of diversity at various levels of ecological organization.

REFERENCES

Aarssen, L. W. (1997). "High productivity in grassland ecosystems: effected by species diversity or productive species?" Oikos **80**(1): 183-184.

Alston, J. M., G. W. Norton, et al. (1998). Science Under Scarcity: Principles and Practice for Agricultural Research Evaluation and Priority Setting. Wallingford, UK, CAB International.

Altieri, M. A. (1991). "How Best Can We Use Biodiversity in Agroecosystems." Outlook on Agriculture **20**(1): 15-23.

Andow, D. A. (1991). "Vegetational Diversity and Arthropod Population Response." Annual Review of Entomology **36**: 561-586.

Armbrecht, I. and I. Perfecto (2003). "Litter-twig dwelling ant species richness and predation potential within a forest fragment and neighboring coffee plantations of contrasting habitat quality in Mexico." Agriculture Ecosystems & Environment **97**(1-3): 107-115.

Benton, T. G., J. A. Vickery, et al. (2003). "Farmland biodiversity: is habitat heterogeneity the key?" Trends in Ecology & Evolution **18**(4): 182-188.

Bommarco, R. (1998). "Reproduction and energy reserves of a predatory carabid beetle relative to agroecosystem complexity." Ecological Applications **8**(3): 846-853.

Bommarco, R. and J. E. Banks (2003). "Scale as modifier in vegetation diversity experiments: effects on herbivores and predators." Oikos **102**(2): 440-448.

Breunig, K. (2003). Losing Ground: At What Cost? Changes in Land Uses and Their Impact on Habitat, Biodiversity, and Ecosystem Services in Massachusetts

Massachusetts Audubon Summary Report.

Briol, E. (2004). Valuing Agricultural Biodiversity on Home Gardens in Hungary: An Application of Stated and Revealed Preference Methods, University College London, University of London. **PhD thesis**.

Bullock, J. M., R. F. Pywell, et al. (2001). "Restoration of biodiversity enhances agricultural production." Ecology Letters **4**(3): 185-189.

Burd, M. (1994). "Bateman Principle and Plant Reproduction - the Role of Pollen Limitation in Fruit and Seed Set." Botanical Review **60**(1): 83-139.

Cane, J. H. and V. J. Tepedino (2001). "Causes and extent of declines among native North American invertebrate pollinators: Detection, evidence, and consequences." Conservation Ecology **5**(1): art. no.-1.

Chapin, F. S., E. S. Zavaleta, et al. (2000). "Consequences of changing biodiversity." Nature **405**(6783): 234-242.

Conway, G. R. (1987). "The Properties of Agroecosystems." Agricultural Systems **24**(2): 95-117.

Costanza, R., R. dArge, et al. (1997). "The value of the world's ecosystem services and natural capital." Nature **387**(6630): 253-260.

Cragg, R. G. and R. D. Bardgett (2001). "How changes in soil faunal diversity and composition within a trophic group influence decomposition processes." Soil Biology and Biochemistry **33**: 2073–2081.

Daily, G. C., G. Ceballos, et al. (2003). "Countryside biogeography of neotropical mammals: Conservation opportunities in agricultural landscapes of Costa Rica." Conservation Biology **17**(6): 1814-1826.

Di Falco, S. and C. Perrings (2003). "Crop genetic diversity, productivity and stability of agroecosystems. A theoretical and empirical investigation." Scottish Journal of Political Economy **50**(2): 207-216.

Drake, L. (1992). "The non-market value of the Swedish agricultural landscape." European Review of Agricultural Economics **19**: 351-364.

Drucker, A. G., V. Gomez, et al. (2001). "The economic valuation of farm animal genetic resources: a survey of available methods." Ecological Economics **36**(1): 1-18.

Ehrlich, P. R. and E. O. Wilson (1991). "Biodiversity Studies - Science and Policy." Science **253**(5021): 758-762.

Evenson, R. E. and D. Gollin (2003). "Assessing the impact of the Green Revolution, 1960 to 2000." Science **300**(5620): 758-762.

Eviner, V. T. and F. S. Chapin (2003). Biogeochemical interactions and biodiversity. Interactions of the Major Biogeochemical Cycles--Global Change and Human Impacts. J. M. Melillo, C. B. Fileld and B. Moldan. Washington, D.C., Island Press**:** 151-173.

Fleischer, A. and Y. Tsur (2000). "Measuring the recreational value of agricultural landscape." European Review of Agricultural Economics **27**(3): 385-398.

Folke, C., J. Colding, et al. (2003). Synthesis: Building resilience and adaptive capacity in social-ecological systems. Navigating Social-Ecological Systems: Building Resilience for Complexity and Change. F. Berkes, J. Colding and C. Folke. Cambridge, UK., Cambridge University Press**:** 352-387.

Freckman, D. W. and R. A. Virginia (1997). "Low-diversity Antarctic soil nematode communities: Distribution and response to disturbance." Ecology **78**(2): 363-369.

Gandini, G. C. and E. Villa (2003). "Analysis of the cultural value of local livestock breeds: a methodology." Journal of Animal Breeding and Genetics **120**(1): 1-11.

Gitay, H., J. B. Wilson, et al. (1996). "Species redundancy: A redundant concept?" Journal of Ecology **84**(1): 121-124.

Griffiths, B. S., K. Ritz, et al. (2000). "Ecosystem response of pasture soil communities to fumigation-induced microbial diversity reductions: an examination of the biodiversity-ecosystem function relationship." Oikos **90**(2): 279-294.

Hector, A., B. Schmid, et al. (1999). "Plant diversity and productivity experiments in European grasslands." Science **286**(5442): 1123-1127.

Heisey, P. W., M. Smale, et al. (1997). "Wheat rusts and the cost of genetic diversity in the Punjab of Pakistan." American Journal of Agricultural Economics **79**(3): 726-737.

Heywood, V. H. (1995). Global Biodiversity Assessment. Cambridge, Cambridge University Press.

Holling, C. S. (1973). "Resilience and stability of ecological systems. Annual Review of Ecology and Systematics " Annual Review of Ecology and Systematics **4**: 1-23.

Holling, C. S., D. S. Schindler, et al. (1995). Biodiversity in the functioning of ecosystems: An ecological synthesis. . Biodiversity Loss: Economic and Ecological Issues. C. Perrings, K.-G. Mäler, C. Folke, C. S. Holling and B.-O. Jansson. Cambridge, U.K., Cambridge University Press**:** 44-83.

Honey, M. (1999). Ecotourism and Sustainable Development: Who Owns the Paradise? . Washington, D.C., Island Press.

Hooper, D. U. and P. M. Vitousek (1997). "The effects of plant composition and diversity on ecosystem processes." Science **277**(5330): 1302-1305.

Huston, M. A. (1997). "Hidden treatments in ecological experiments: Re-evaluating the ecosystem function of biodiversity." Oecologia **110**(4): 449-460.

Jones, C. G., J. H. Lawton, et al. (1994). "Organisms as Ecosystem Engineers." Oikos **69**(3): 373-386.

Kinzig, A. P., S. W. Pacala, et al., Eds. (2002). Functional consequences of biodiversity : empirical progress and theoretical extensions, Princeton University Press.

Koo, B., P. G. Pardey, et al. (2004). Saving Seeds: The Economics of Conserving Crop Genetic Resources Ex Situ in the Future Harvest Centres of the CGIAR. Wallingford, UK, CABI Publishing.

Kremen, C. and T. Ricketts (2000). "Global perspectives on pollination disruptions." Conservation Biology **14**(5): 1226-1228.

Kremen, C., N. M. Williams, et al. (2002). "Crop pollination from native bees at risk from agricultural intensification." Proceedings of the National Academy of Sciences of the United States of America **99**(26): 16812-16816.

Kumar, B. M. and P. K. R. Nair (2004). "The enigma of tropical homegardens." Agroforestry Systems **61**(1): 135-152.

Loreau, M., S. Naeem, et al., Eds. (2002). Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. . Oxford, UK, Oxford University Press.

Mader, P., A. Fliessbach, et al. (2002). "Soil fertility and biodiversity in organic farming." Science **296**(5573): 1694-1697.

McNeely, J. A. and S. J. Scherr (2003). *Ecoagriculture: Strategies to Feed the World and Save Biodiversity*. Washington, dc: Island Press. Washington, D.C., Island Press.

Meng, E. C. H., M. Smale, et al. (2003). Wheat genetic diversity in China: Measurement and cost. . Agricultural Trade and Policy in China: Issues, Analysis and Implications S. Rozello and D. A. Sumner. Burlington, VT., Ashgate.

Michon, G. and H. de Foresta (1990). Complex agroforestry systems and the conservation of biological diversity. Agroforests in Indonesia: The link between two worlds. . the International Conference on Tropical Biodiversity, Kuala Lumpur, Malaysia., United Selangor Press.

Mikola, J. and H. Setala (1998). "No evidence of trophic cascades in an experimental microbial-based soil food web." Ecology **79**(1): 153-164.

Milestad, R. and S. Hadatsch (2003). "Organic farming and social-ecological resilience: the alpine valleys of Solktaler, Austria." Conservation Ecology **8**(1): 18.

Montoya, J. M., M. A. Rodriguez, et al. (2003). "Food web complexity and higher-level ecosystem services." Ecology Letters **6**(7): 587-593.

Mulder, C. P. H., J. Koricheva, et al. (1999). "Insects affect relationships between plant species richness and ecosystem processes." Ecology Letters **2**(4): 237-246.

Nabhan, G. P. and S. Buchmann (1997). Services provided by pollinators. . Nature's Services. Societal Dependence on Natural Ecosystems. G. C. Daily. Washington, D.C., Island Press: 133-150.

Naeem, S. (1998). "Species redundancy and ecosystem reliability." Conservation Biology **12**(1): 39-45.

Naeem, S. (2001). Experimental validity and ecological scale as tools for evaluating research programs. . Scaling Relationships in Experimental Ecology. R. H. Gardner, W. M. Kemp, V. S. Kennedy and J. E. Petersen. New York, Columbia

University Press**:** 223-250.

Naeem, S. (2002). "Ecosystem consequences of biodiversity loss: The evolution of a paradigm." Ecology **83**(6): 1537-1552.

Naeem, S., D. R. Hahn, et al. (2000). "Producer-decomposer co-dependency influences biodiversity effects." Nature **403**(6771): 762-764.

Naeem, S. and S. B. Li (1997). "Biodiversity enhances ecosystem reliability." Nature **390**(6659): 507-509.

Naeem, S., L. J. Thompson, et al. (1994). "Declining Biodiversity Can Alter The Performance Of Ecosystems." Nature **368**(6473): 734-737.

Naeem, S., L. J. Thompson, et al. (1995). "Empirical-Evidence That Declining Species-Diversity May Alter the Performance of Terrestrial Ecosystems." Philosophical Transactions of the Royal Society of London Series B-Biological Sciences **347**(1321): 249-262.

OECD (2002). Handbook of biodiversity valuation-a guide for policy makers, OECD.

Orians, G. H., G. M. Brown, et al., Eds. (1990). Preservation and Valuation of Biological Resources. Seattle, WA., University of Washington Press.

Ostman, O., B. Ekbom, et al. (2001). "Landscape complexity and farming practice influence the condition of polyphagous carabid beetles." Ecological Applications **11**(2): 480-488.

Pacala, S. W. and D. Tilman (2002). The transition from sampling to complementarity. Functional consequences of biodiversity : empirical progress and theoretical extensions. A. P. Kinzig, S. W. Pacala and D. Tilman. Princeton, NJ, Princeton University Press**:** 151-166.

Pain, D. J. and M. W. Pienkowski (1997). Farming and Birds in Europe: The Common Agricultural Policy and Its Implications for Bird Conservation. . Cambridge, UK, Academic Press.

Paine, R. T. (1966). "Food web complexity and species diversity." American Naturalist **100**: 65-75.

Perfecto, I., J. Vandermeer, et al. (1997). "Arthropod biodiversity loss and the transformation of a tropical agro-ecosystem." Biodiversity and Conservation **6** 935-945.

Perrings, C. (1995). Biodiversity conservation as insurance. The economics and ecology of biodiversity. T. Swanson. New York, Cambridge University Press**:** 91-98.

Petchey, O. L., P. J. Morin, et al. (2002). Contributions of aquatic model systems to our understanding of biodiversity and ecosystem functioning. Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. . M. Loreau, S. Naeem and P. Inchausi. Oxford, UK, Oxford University Press**:** 127-138.

Pinto-Correia, T. (2000). "Future development in Portuguese rural areas: how to manage agricultural support for landscape conservation?" Landscape and Urban Planning **50**(1-3): 95-106.

Pretty, J. and H. Ward (2001). "Social capital and the environment." World Development **29**(2): 209-227.

Ricketts, T. H., G. C. Daily, et al. (2001). "Countryside biogeography of moths in a fragmented landscape: Biodiversity in native and agricultural habitats." Conservation Biology **15**(2): 378-388.

Robinson, R. A., J. D. Wilson, et al. (2001). "The importance of arable habitat for farmland birds in grassland landscapes (vol 38, pg 1059, 2001)." Journal of Applied Ecology **38**(6): 1386-1386.

Schimid, B., J. Joshi, et al. (2002). Empirical evidence for biodiversity–ecosystem functioning relationships. Functional consequences of biodiversity : empirical progress and theoretical extensions. A. P. Kinzig, S. W. Pacala and D. Tilman, Princeton University Press**:** 120-150.

Scoones, I. (1999). "New ecology and the social sciences: What prospects for a fruitful engagement?" Annual Review of Anthropology **28**: 479-507.

Smale, M., Ed. (2005). Valuing Crop Biodiversity: On-Farm Genetic Resources and Economic Change. Wallingford, UK CABI Publishing.

Smale, M., J. Hartell, et al. (1998). "The contribution of genetic resources and diversity to wheat production in the Punjab of Pakistan." American Journal of Agricultural Economics **80**(3): 482-493.

Soderstrom, B., S. Kiema, et al. (2003). "Intensified agricultural land-use and bird conservation in Burkina Faso." Agriculture Ecosystems & Environment **99**(1-3): 113-124.

Southwick, E. E. and L. Southwick (1992). "Estimating the Economic Value of Honey-Bees (Hymenoptera, Apidae) as Agricultural Pollinators in the United-States." Journal of Economic Entomology **85**(3): 621-633.

Steffan-Dewenter, I. (2003). "Importance of habitat area and landscape context for species richness of bees and wasps in fragmented orchard meadows." Conservation Biology **17**(4): 1036-1044.

Steffan-Dewenter, I., U. Munzenberg, et al. (2002). "Scale-dependent effects of landscape context on three pollinator guilds." <u>Ecology</u> **83**(5): 1421-1432.

Stein, T. V., D. H. Anderson, et al. (1999). "Using stakeholders' values to apply ecosystem management in an upper midwest landscape." <u>Environmental Management</u> **24**(3): 399-413.

Tews, J., U. Brose, et al. (2004). "Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures." <u>Journal of Biogeography</u> **31**(1): 79-92.

Thies, C. and T. Tscharntke (1999). "Landscape structure and biological control in agroecosystems." <u>Science</u> **285**(5429): 893-895.

Thrupp, L. A. (1997). Linking Biodiversity and Agriculture: Challenges and Opportunities for Sustainable Food Security. Washington, DC, World Resource Institute.

Tilman, D., J. Knops, et al. (2002). Experimental and observational studies of diversity, productivity, and stability. <u>Functional consequences of biodiversity : empirical progress and theoretical extensions</u>. A. P. Kinzig, S. W. Pacala and D. Tilman, Princeton University Press**:** 42-70.

Tilman, D., J. Knops, et al. (1997). "The influence of functional diversity and composition on ecosystem processes." <u>Science</u> **277**(5330): 1300-1302.

Tilman, D., C. L. Lehman, et al. (1997). "Plant diversity and ecosystem productivity: Theoretical considerations." <u>Proceedings of the National Academy of Sciences of the United States of America</u> **94**(5): 1857-1861.

Tilman, D., D. Wedin, et al. (1996). "Productivity and sustainability influenced by biodiversity in grassland ecosystems." <u>Nature</u> **379**(6567): 718-720.

Tonhasca, A. and B. R. Stinner (1991). "Effects of Strip Intercropping and No-Tillage on Some Pests and Beneficial Invertebrates of Corn in Ohio." Environmental Entomology **20**(5): 1251-1258.

Turner, R. K., J. Paavola, et al. (2003). "Valuing nature: lessons learned and future research directions." Ecological Economics **46**(3): 493-510.

Walker, B., S. Carpenter, et al. (2002). "Resilience management in social-ecological systems: a working hypothesis for a participatory approach." Conservation Ecology **6**(1).

Walker, B. H. (1992). "Biodiversity and Ecological Redundancy." Conservation Biology **6**(1): 18-23.

Walker, D. J. and D. L. Young (1986). "The Effect of Technical Progress on Erosion Damage and Economic Incentives for Soil Conservation." Land Economics **62**(1): 83-93.

Wardle, D. A., O. Zackrisson, et al. (1997). "The influence of island area on ecosystem properties." Science **277**(5330): 1296-1299.

Widawsky, D. and S. Rozelle (1998). Varietal diversity and yield variability in Chinese rice production. . Farmers, Gene Banks, and Crop Breeding. M. Smale. Boston, Kluwer.

Wilson, E. O. (1988). Biodiversity. Washington, D.C., National Academy Press.

Wood, S. K. and S. J. Scherr (2000). Pilot Analysis of Global Ecosystems: Agroecosystems. Washington, DC: International Food Policy Research Institute and World Resources Institute. Washington, DC, International Food Policy Research Institute and World Resources Institute.

Wright, J. P., C. G. Jones, et al. (2002). "An ecosystem engineer, the beaver, increases species richness at the landscape scale." Oecologia **132**(1): 96-101.

Yachi, S. and M. Loreau (1999). "Biodiversity and ecosystem productivity in a fluctuating environment: The insurance hypothesis." Proceedings of the National Academy of Sciences of the United States of America **96**(4): 1463-1468.

Zhu, Y. Y., H. R. Chen, et al. (2000). "Genetic diversity and disease control in rice." Nature **406**(6797): 718-722.

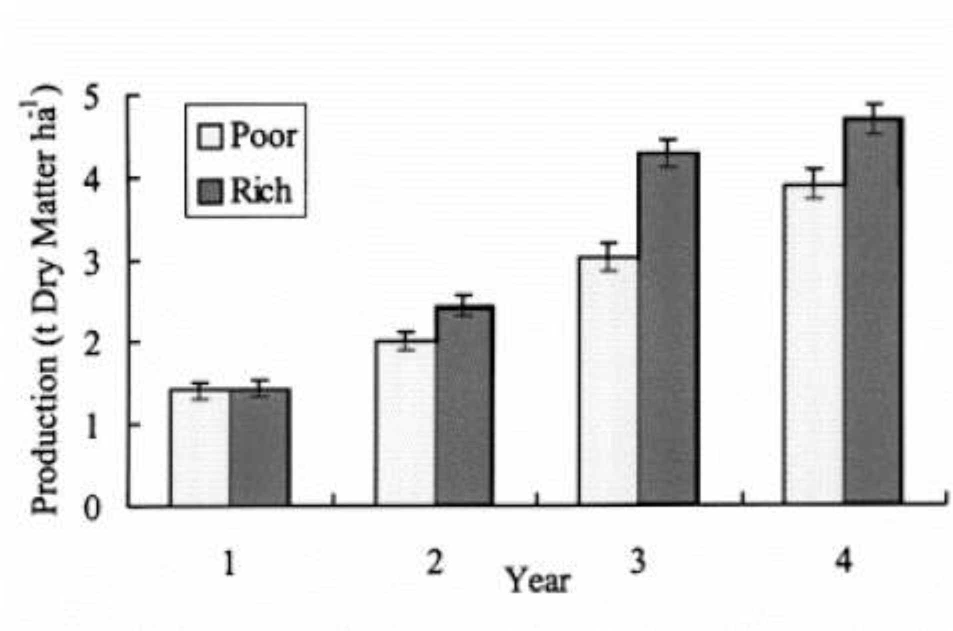

Figure 1. Biodiversity treatment effects on hay production in different years (mean across plots and sites is displayed ± one standard error). The species-rich treatment had higher dry matter yield from the second year onward. Adapted from Bullock et al. 2001.

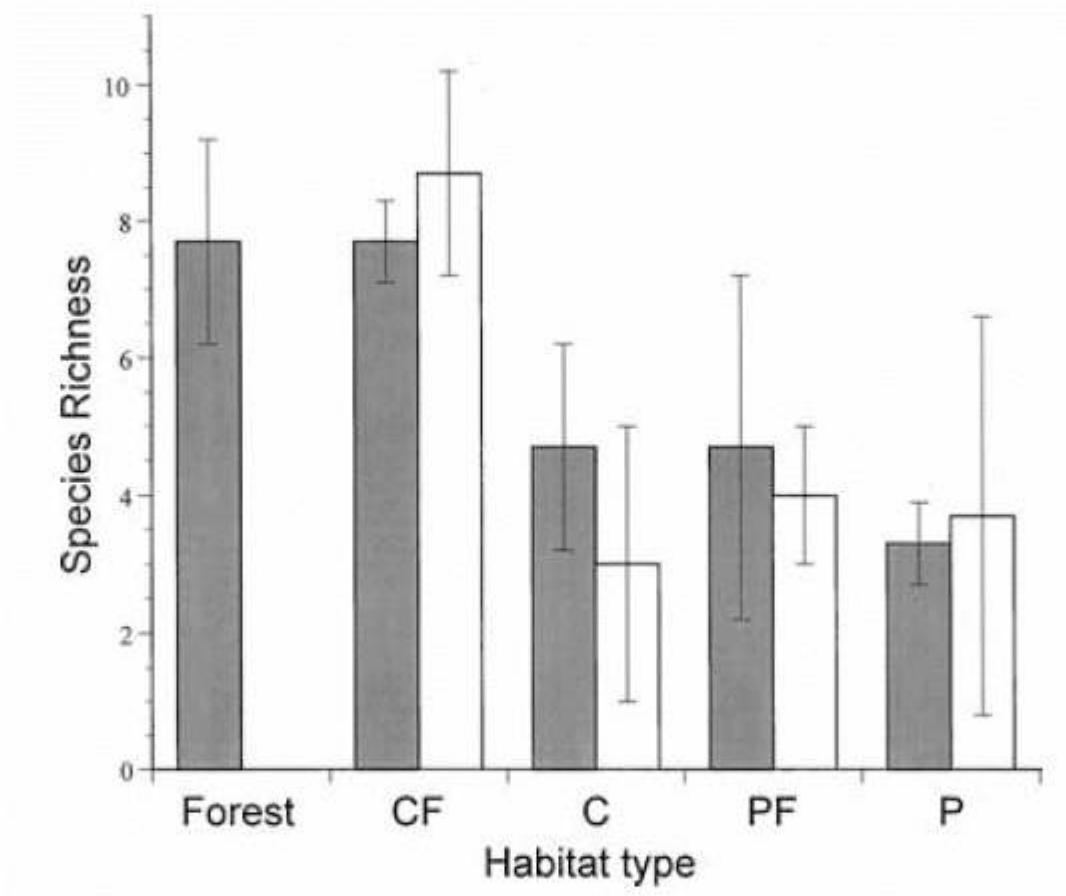

Figure 2. Mammal species richness by habitat type and distance class from an extensive forest patch (mean ± one standard error). Shaded bars represent sites in and near (<1km) the forest; open bars represent sites far from (5-7 km) the forest. Species richness varied significantly among habitat types but not with distance from extensive forest.
Small forest remnants contiguous with coffee plantations (CF) did not differ from more extensive forest in species richness and were richer than coffee plantations (C), pastures with adjacent forest remnant (PF), and pastures (P). Adapted from Daily et al. 2003.

Table 1. Summary of average global value of annual ecosystem services. Numbers in the body of the table are in \$ ha^{-1} yr^{-1}. Rows and column totals are in \$ yr^{-1} x 10^9, column totals are the sum of the products of the per ha services in the table and the area of each biome. a = Total value per ha is in \$ ha^{-1} yr^{-1}; b= Total global flow value is in \$ yr^{-1} x 10^9. Shaded cells indicate services that do not occur or are known to be negligible. Open cells indicate lack of available information. Adapted from Costanza et al. 1997.

		Ecosystem services (1994 US$ ha^{-1} yr^{-1})																		
Biome	**Area**	Gas regulation	Climate regulation	Disturbance regulation	Water regulation	Water supply	Erosion control	Soil formation	Nutrient cycling	Waste treatment	Pollination	Biological control	Habitat/refugia	Food production	Raw materials	Genetic resources	Recreation	Cultural	Total Value per	To
Marine	36302																		577	20949
Open Ocean	33200	38							118			5		15	0			76	252	8381
Coastal	3102			88					3677			38	8	93	4		82	62	4052	12568
Estuaries	180			567					21100			78	131	521	25		381	29	22832	4110
Seagrass/A	200								19002						2				19004	3801
Coral Reefs	62			2750						58		5	7	220	27		3008	1	6075	375
Shelf	2,660								1431			39		68	2			70	1610	4283
Terrestrial	15323																		804	12319
Forest	4855		141	2	2	3	96	10	361	87		2		43	138	16	66	2	969	4706
Tropical	1900		223	5	6	8	245	10	922	87				32	315	41	112	2	2007	3813
Temperate	2955		88		0			10		87		4		50	25		36	2	302	894
Grasslands/R	3898	7	0		3		29	1		87	25	23		67		0	2		232	906
Wetlands	330	133		4539	15	3800				4177			304	256	106		574	881	14785	4879
Tidal marsh	165			1839						6696			169	466	162		658		9990	1648
Swamps/Fl	165	265		7240	30	7600				1659			439	47	49		491	1761	19580	3231
Lakes/Rivers	200				5445	2117				665				41			230		8498	1700
Deserts	1925																			
Tundra	743																			
Ice/Rock	1640																			
Cropland	1400										14	24		54					92	128

Biodiversity and Ecosystem Services: A multi-scale empirical study of the relationship between species richness and net primary production[*]

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Abstract

Biodiversity (BD) and Net Primary Productivity (NPP) are intricately linked in complex ecosystems such that a change in the state of one of these variables can be expected to have an impact on the other. Using multiple regression analysis at the site and ecoregion scales in North America, we estimated relationships between BD (using plant species richness as a proxy) and NPP (as a proxy for ecosystem services). At the site scale, we found that 57% of the variation in NPP was correlated with variation in BD after effects of temperature and precipitation were accounted for. At the ecoregion scale, 3 temperature ranges were found to be important. At low temperatures (-2.1ºC average) BD was negatively correlated with NPP. At mid temperatures (5.3ºC average) there was no

[*] This paper was published in *Ecological Economics* **61**(2-3): 478-491.

correlation. At high temperatures (13ºC average) BD was positively correlated with NPP, accounting for approximately 26% of the variation in NPP after effects of temperature and precipitation were accounted for. The general conclusion of positive links between BD and ecosystem functioning from earlier experimental results in micro and mesocosms was qualified by our results, and strengthened at high temperature ranges. Our results can also be linked to estimates of the total value of ecosystem services to derive an estimate of the value of the biodiversity contribution to these services. We tentatively conclude from this that a 1% change in BD in the high temperature range (which includes most of the world's BD) corresponds to approximately a 1/2% change in the value of ecosystem services.

Keywords: biodiversity, net primary production, ecosystem services, species richness

Introduction

Biodiversity is the variability among living organisms from all sources. This includes diversity within species, between species and of ecosystems (Heywood 1995). In the past 100 years biodiversity loss has been so dramatic that it has been recognized as a global change in its own right (Walker and Steffen 1996). This has raised numerous concerns, including the possibility that the functioning of earth's ecosystems might be threatened by biodiversity loss (Ehrlich and Ehrlich 1981; Schulze and H.A. 1993)

Ecosystem functions refer variously to the habitat, biological or system properties, or processes of ecosystems. Ecosystem goods (such as food) and services (such as waste assimilation) represent the benefits human populations derive, directly or indirectly, from ecosystem functions (Costanza, dArge et al. 1997). If biodiversity has an influence on ecosystem functioning (in addition to any other roles it may play) then it will affect ecosystem goods and services and human welfare. Research on the relationship between biodiversity and ecosystem functioning (BDEF) is therefore of direct relevance to public policy, and this relationship has been the subject of considerable interest and controversy over the past decade (Cameron 2002).

The relationship between biodiversity and ecosystem functioning has historically been a central concern of ecologists. But the direction and underlying mechanisms of this relationship has been a topic of ongoing controversy, which has been complicated by the many different types (e.g. species, genetic, community, functional) and measures (e.g. richness, evenness, Shannon-Weaver) of diversity. The discussion has also been complicated because in the public policy arena, the term biodiversity is often erroneously equated with the totality of life, rather than its variability.

In 1972 Robert May, using linear stability analysis on models based on randomly constructed communities with randomly assigned interaction strengths, found that in general diversity tends to destabilize community dynamics (May 1972). This result was at odds with the earlier hypotheses (Odum 1953; MacArthur 1955; Elton 1958) that diversity leads to increased productivity and stability in ecological communities.

Recent studies have attempted to understand the effects of diversity on ecosystem functioning using experimental ecosystems, including microcosms (Naeem, Thompson et al. 1994; Naeem, Hakansson et al. 1996) and grassland mesocosms (Naeem, Thompson et al. 1994; Tilman and Downing 1994; Naeem, Hakansson et al. 1996; Tilman, Wedin et al. 1996; Tilman, Knops et al. 1997). These studies seem to provide experimental evidence for a positive relationship between biodiversity and ecosystem functioning in general, and between biodiversity and NPP in particular (Naeem, Thompson et al. 1995; Tilman, Wedin et al. 1996; Tilman, Knops et al. 1997; Lawton 1998). However, some have argued that the micro and mesocosm experiments showed no "real" effect of biodiversity because the results of these experiments were only due to "sampling effect" artifacts of the way the experiments were conducted (Aarssen 1997; Grime 1997; Huston 1997; Wardle, Zackrisson et al. 1997).

The debate continues. Recent experimental studies have claimed various relationships such as increases in biodiversity positively affecting productivity but decreasing stability (Pfisterer and Schmid 2002); increases in biodiversity increasing productivity but only due to one or two highly productive species (Paine 2002); and Willms (2002) suggests that there is no general relationship between these two factors due to species specific effects and unique trophic links. Further, Wardle and

Zackrisson's (2005) studies on island ecosystems found that effect of biotic losses on ecosystem functions depends greatly on individual biotic and abiotic characteristics of the system.

Obviously, the links between biodiversity and ecosystem functioning are complex, and it should come as no surprise that simple answers have not emerged. It is also the case that small scale, short duration micro and mesocosm experiments (while attractive because they are the only controlled experiments that can reasonably be done on these questions) cannot necessarily be directly extrapolated to the real world. These short-term, small-scale experiments rely on communities that are synthesized from relatively small species pools and in which conditions are highly controlled. Practical limitations simply preclude controlled experiments that can span the large spatial scales, the long temporal scales, and the representative diversity and environmental gradients that are properly the concern of work in this area. This limits our ability to directly extrapolate the results of small-scale experiments to longer time scales and larger spatial scales (Symstad, Chapin et al. 2003). Additional information on larger scales is thus essential in informing the debate about the interpretation of experiments designed to examine the relationship between biodiversity and ecosystem functioning and services, and the applicability of those experiments to the "real world" (Kinzig and S.W. 2002).

Part of the fuel for the ongoing debate on the subject, is the fact that biodiversity is both a cause of ecosystem functioning and a response to changing conditions (Hooper, Chapin et al. 2005). The components of complex ecological systems, like those investigated in the BDEF relationship, also operate at different but overlapping spatial and temporal scales (Limburg, O'Neill et al. 2002). The assumption that causal chains operate on one temporal and spatial scale at a time is inconsistent with what we know about ecological systems (Allen and Starr 1982). Rather than a linear additive process, complex systems are defined by feedback loops, blurring the distinction between cause and effect. This blurring of cause and effect contributes to the BDEF debate.

In this paper we try to address the BDEF relationship while leaving the 'prime mover' discussion aside. Our investigation specifically looks at the relationship between NPP and vascular plant diversity (hereon biodiversity or BD). This relationship is likely characterized by the following simultaneous causal links:

- NPP responding to temperature, precipitation, soil characteristics and other abiotic factors
- BD responding to temperature, precipitation, soil characteristics and other abiotic factors
- NPP responding to BD
- BD responding to NPP

The very nature of ecological systems forces us to consider these multiple relationships between NPP and BD. Assuming temperature and precipitation (as well as other determinants of system productivity) are positive antecedents of both BD and NPP, the relationship between BD and NPP can be characterized as one of the following (figure 1):

INSERT Figure 1

In Case 1, the positive relationship between BD and NPP is amplified by the anteceding influence of temperature and precipitation. If this were the case, we would predict that the bivariate coefficient of variation between NPP and BD should be greater (in absolute value) than the partial correlation coefficient, controlling for temperature and precipitation. In Case 2, the negative relationship between BD and NPP is suppressed by the abiotic influences. In this case, the partial correlation coefficient would be more (in absolute value) than the bivariate coefficient between NPP and BD. Note that nothing in this analysis assumes causality. The arrow between BD and NPP could also go in the other direction.

In order to address this relationship we synthesized empirical data at the site and eco-region scales. Recent advances in the availability of biodiversity and NPP data have made this synthesis possible.

Methods

Biodiversity takes many forms (e.g. genetic, functional, and landscape diversity) in addition to simple species richness (Tilman and Lehman 2002). However, measurements of these other aspects are in general not available at large scales, and the number of species has been the focus of most of the recent research on the BDEF relationship. We therefore used species richness as a (admittedly imperfect) proxy for biodiversity. Within this, we focused on vascular plant species richness because it was both available at both of our scales of interest and most directly relevant to NPP.

There is a long list (Costanza, dArge et al. 1997; de Groot, Wilson et al. 2002) of ecosystem services, but there is limited data on most of them. However, aboveground net primary production (NPP) data are available at multiple scales and NPP has been shown to correlate with the total value of ecosystem services (Costanza, d'Arge et al. 1998). NPP measurements are also widely employed in BDEF research at the micro- and mesocosm scales. In addition, NPP is commonly used as an index to reflect ecosystem response to climate change (McCarthy and Intergovernmental Panel on Climate Change. Working Group II. 2001). In general, aboveground NPP is much more readily available than total (above and below ground) NPP, so we used aboveground NPP for this study.

For the "site" scale of analysis (Scale 1) we performed an extensive literature search using the ISI Web of Knowledge and other tools (i.e. library-based bibliographic search engines) and were able to obtain approximately 200 observations on NPP from a total of 52 spatial locations globally. However, we found no observational studies that directly measured both NPP and total plant diversity simultaneously at specific locations. For the most part, the studies we encountered were species-specific, linking limited groups of species to NPP. Therefore, we were forced to search for data on biodiversity, environmental variables, and NPP separately, with spatial location as the key link among these data. Long-Term Ecological Research (LTER) and Forest Service research sites in North America were the only sites for which the required data were available (Knapp and Smith 2001). Although limited in number, these sites span a wide range geographically and biophysically from temperate forests, to tundra to high mountain meadows. For NPP data in our Scale 2 (ecoregion) analysis we used recent global NPP satellite derived estimates, as explained below.

Biodiversity data were the main variable of interest for the study and also the most difficult to standardize across sites. Our search revealed numerous gaps in the literature for biodiversity counts in spite of the increasing effort within the field to develop more accurate biodiversity figures. For our Scale 1 analysis, a few sites had biodiversity counts for the site, but not necessarily from the exact plots where the NPP data was derived. While this is a limitation, it is a bias that applies to all sites equally. The sites for which some information for both NPP and biodiversity was available was limited to 11 usable sites. Obtaining better biodiversity data for additional sites for which NPP measurements are ongoing could greatly expand the number of usable data points. For Scale 2, we used the work on North American Ecoregions of Ricketts et al. (Ricketts and Dinerstein 1999) on biodiversity by ecoregion.

In addition to biodiversity, several physical environmental factors are important in explaining variations in ecosystem functions and services across sites. Temperature, precipitation, and soil organic matter content are three such factors we were able to include in this analysis. Temperature and precipitation have long been known to explain much of the basic global pattern of NPP (Lieth 1978). Precipitation and temperature data were obtained from the Global Climate Database (Leemans and Cramer 1991). Station data were extrapolated to create a full-coverage map for the entire United States in order to estimate the values for each of our sites.

We determined the soil type at each site using the FAO Digital Soil Map of the World (1995) and the latitudes and longitudes of the study sites. The FAO map yielded two useful figures for organic carbon content; the percent organic carbon of the topsoil and the percent organic content of the subsoil. The first thirty centimeters of soil was considered topsoil, while 30cm to 100cm was considered to be subsoil. Weighted averages were calculated when different horizons were present.

Scale 1: Site Level Analysis

Table 1 is a list of all the data used in the regression analysis of NPP with biodiversity and physical characteristics at the site scale. Step-wise regression was used to determine the most significant determinants of NPP over the entire data set. BD was incorporated untransformed and log-transformed. Step-wise regression yielded the following as the best model:

$$NPP = \alpha + \beta_1 * P + \beta_2 * BD + \beta_3 * \ln(BD)$$

NPP = Aboveground Net Primary Production

BD = vascular plant species number

P = growing season precipitation

Temperature, and organic carbon content proved not to be significant explanatory variables at this scale.

All predictors were tested for suitably normal distributions using Q-normal plots. Tolerances were calculated for each of the predictor variables to test for collinearity. Tolerance for the biodiversity terms was only 0.09 suggesting a high level of collinearity. However, neither term was significant alone implying a nonlinear relationship. We recalculated the coefficients using a generalized linear model that showed the coefficient estimates to not be biased.

Table 2 shows the Ordinary Least Squares (OLS) regression coefficients for this model.

[INSERT TABLE 2]

R^2 for the model was 0.85 with p = 0.0011. The squared partial correlation for the two BD terms controlling for temperature and precipitation reveals that 57% of the variation in NPP was correlated with variation in BD, though with such a small number of data points this figure has a low statistical power. Using the regression model, we can calculate the partial derivative of NPP with respect to BD:

$$\partial NPP / \partial BD = 0.857 - \frac{542.9}{BD}.$$

For 8 out of 12 sites, this yields a negative correlation between marginal NPP and marginal BD, with influence becoming increasingly negative with lower diversity. This equation implies that the marginal rate of change of NPP with BD increases with increasing BD.

Scale 2: North American Eco-Region Analysis

Ecoregions are defined as a physical area having similar environmental/geophysical conditions as well as a similar assemblage of natural communities and ecosystem dynamics. North America has been divided into 116 eco-regions for which data has been assembled for several types of biological diversity (including vascular plant, tree species, snails, butterflies, birds, and mammals), geophysical characteristics, and habitat threats (Ricketts and Dinerstein 1999).

The Numerical Terradynamic Simulation Group (NTSG), at the University of Montana used MODIS 1 km^2 resolution satellite imagery from 2001 coupled with parameters derived from the Biome-BGC, a globalized version of the Forest-BGC model (Running and Coughlan 1988; Turner, Ritts et al. 2003), to estimate NPP as a function of Leaf Area Index (LAI), Fractional Photosynthetically Active Radiation (FPAR), temperature, precipitation and soil properties. Eight-day estimates of NPP are averaged over an entire year (2001, in this case), correcting for seasonal variation. Explicit details concerning the algorithms used to derive NPP estimates can be found at the NTSG website at: http://www.ntsg.umt.edu.

Due to the size of this dataset, we resampled the 1 km^2 MODIS/NTSG data to 10 km^2 resolution using a nearest neighbor interpolation method. Global land cover data was obtained from the United Nations Environment Network website at: http://www.unep.net/ . This data was derived from AVHRR satellite data (1 km resolution) and was classified into 19 land cover categories. NPP values that were labeled *crop, urban, barren, ice* or *water*, were removed from the analysis. NPP values for agricultural areas were removed from the analysis because it was expected that high fertilizer and irrigation inputs to these lands would boost NPP estimates but have a negative effect on biodiversity, thus reducing the relationship between NPP and biodiversity for intensively managed or altered lands. Therefore the aggregate area included in the analysis is loosely defined as 'natural area.' The remaining NPP values were then aggregated by eco-region to produce estimates of the average annual aboveground NPP for North American eco-regions for the year 2001. From this combination of sources we obtained data for 102 ecoregions for the following parameters: Number of Vascular Plants per 10,000 km^2 (hereafter BD for biodiversity), Net Primary Production (NPP), Mean Annual Precipitation (P), and Mean Annual Temperature (T). These data are listed in Supplementary Table S1.

While it would have been preferable to use direct measurements of NPP rather than modeled data based on remote sensing images, this was not an option. Further, since temperature and precipitation are drivers of both NPP and plant diversity, it is critical that they be incorporated in our model despite the fact that these parameters were also used to derive the NPP estimates.

Step-wise regression was used to determine the most significant determinants of NPP over the entire data set. Precipitation was log-transformed and BD was incorporated untransformed and log-transformed. Step-wise regression yielded the following as the best model:

$$NPP = \alpha + \beta_1 * T + \beta_2 * \ln(P) + \beta_3 * BD + \beta_4 * \ln(BD)$$

All predictors were tested for suitably normal distributions using Q-normal plots. Tolerances were calculated for each of the predictor variables to test for collinearity. All tolerances were high except for BD, which had a tolerance of 0.28. Since the threshold of inappropriately high collinearity is generally set between 0.20 and 0.25, we retained the parameter. By including both BD and ln(BD), we are able to model a more non-linear relationship between BD and NPP, a strategy that is supported by the site-scale results above. Table 3 shows the Ordinary Least Squares (OLS) regression coefficients for this model.

[INSERT Table 3]

R^2 for the model was 0.58 with $p < 0.0001$. The squared partial correlation for the two BD terms controlling for temperature and precipitation was calculated to be 0.10 implying that BD accounted for 10% of the variation in NPP, assuming this causal direction. Using the regression model, we can calculate the partial derivative of NPP with respect to BD:

$$\partial NPP \big/ \partial BD = 0.159 - \frac{103.7}{BD}.$$

For the vast majority of ecoregions, this yields a negative correlation between marginal NPP and marginal BD, with influence becoming increasingly negative with lower temperature (Figure 2).

However, further exploration using stepwise regression revealed a significant interaction between ln(BD) and temperature. This led us to hypothesize a variation in the relationship between NPP and BD over a temperature gradient.

[Insert Figure 2]

To assess this, we performed the following analysis. First, we ordered the ecoregions by mean annual temperature. Then using the model:

$$NPP = \alpha + \beta_1 * T + \beta_2 * \ln(P) + \beta_3 * \ln(BD),$$

We performed OLS regression using a moving window of 20 data points. We began with the 20 coldest ecoregions, and after each regression moved the window one data point in the direction of higher temperature. This yielded 83 individual regression outputs from which we took the R^2 measure of goodness of fit and the estimated coefficient for ln(BD). We also calculated the average of temperature for all twenty data points in each subset. Finally, we plotted the goodness of fit and the coefficient for ln(BD) as a function of average temperature (Figure 3).

[Insert Figure 3]

Two patterns are apparent. First is the strong dependence of the coefficient of ln(BD) on temperature. Here there are three modes of behavior: consistently negative at low temperatures, consistently positive at high temperatures, and a strong linear trend from low to high at mid-range temperatures. Further, there appear to be two abrupt transition points that demarcate the boundaries between these modes, one at about 2 degrees C and the other around 8 degrees C. Goodness of fit on the other hand follows a V-shaped trend. Fit is fairly high at low and high temperatures, but low at mid-range temperatures, approaching zero at an average temperature of 2.5 degrees C. It is logical that the model should express the weakest fit in the same range at which ln(BD) has the most indeterminate relationship to NPP.

Based on the output in Figure 3 we divided the data set into three subsets with an overlap of 10 data points to account for the scale of the moving window regression. Thus the three subsets are data points 1 – 45 (low temperature range), 35 – 61 (mid-temperature range) and 51 – 102 (high temperature range). The subsets had an average mean annual temperature of -2.1, 5.3, and 13.0 degrees Celsius respectively. Stepwise regression was used to determine the best model in all three ranges with the following results.

Low Temperature

At low temperatures, the mean summer temperature (ST) explains the vast majority of variation in NPP at the ecoregional scale ($R^2 \sim 0.53$). Further, neither BD nor ln(BD) were significant alone, but together they greatly improved the model. All other variables, including surprisingly precipitation, were not significant. This yielded the model:

$$NPP = \alpha + \beta_1 * ST + \beta_2 * BD + \beta_3 * \ln(BD).$$

Ordinary Least Squares (OLS) regression coefficients for this model are shown in Table 4.

]INSERT Table 4]

R^2 for the model was 0.65 with $p < 0.0001$. The squared partial correlation for the BD terms controlling for summer temperature was 0.25. Therefore in this analysis 25% of the variation in NPP corresponded to variation in biodiversity. Using the regression model, we can calculate the partial derivative of NPP with respect to BD:

$$\partial NPP / \partial BD = 0.286 - \frac{115.3}{BD}.$$

As with the regression over the entire data set, this is largely negative (Figure 4). Note that the R^2 measure for NPP as a function of BD and ln(BD) alone is only 0.07, significantly less than the squared partial correlation. This is consistent with BD having a suppression effect on NPP where summer temperature has a positive effect on both BD and NPP (Figure 2).

[Insert Figure 4]

Mid Temperature

Stepwise regression over data points 35 – 61 yielded no variables significant at the 0.10 level. Log-transformed annual precipitation was a mediocre predictor of NPP ($R^2 \sim 0.09$).

High Temperature

In the high temperature range, we could not use Summer Temperature (ST) because the tolerance was only 0.10 indicating an unacceptable level of collinearity in the predictor variables. Stepwise regression using all variables but ST yielded the following model:

$$NPP = \alpha + \beta_1 * T + \beta_2 * \ln(P) + \beta_3 * \ln(BD).$$

Ordinary Least Squares (OLS) regression coefficients for this model are shown in Table 5.

[INSERT Table 5]

R^2 for the model was 0.65 with $p < 0.0001$. The squared partial correlation for ln(BD) was 0.26 suggesting that BD accounted for approximately 26% of the variation in

NPP. This is nearly equal to the bivariate correlation for ln(BD) suggesting a minimal influence of temperature upon BD at this range. Indeed, the bivariate correlation between temperature and ln(BD) is only 0.07.

There were three significant outliers in this data set—Queen Charlotte Islands, Northern California Coastal Forests, and the Sonoran Desert. Queen Charlotte Islands had the highest precipitation of all ecoregions in the data set by almost 20%, while the Sonoran Desert had one of lowest. The Northern California Coastal Forests has the second highest rate of NPP. These outliers suggest marginal effects missed by the linearity of the model. When they are removed, goodness of fit increases significantly ($R^2 = 0.72$), but regression coefficients are not much affected.

[Insert Figure 5]

Discussion: the empirical link between BD and NPP.

The results generate a number of discussion points. This investigation implies that the marginal rate of change of NPP with BD increases with increasing BD. While the data at Scale 1 is sparse and difficult to validate, it is worth noting a very similar model was found as at the ecoregion scale with comparable coefficient estimates. It suggests that if additional observations become available, it would be worth looking for a similar pattern of temperature dependency as was discovered at the ecoregion scale.

The number of observations available for Scale 2 provided latitude for a more rigorous statistical investigation. By including both BD and ln(BD), we were able to model a more non-linear relationship between BD and NPP. Obviously the feedback effects between BD and NPP (Hooper, Chapin et al. 2005) force nonlinearities, but these effects are poorly understood.

The moving window regression, with 83 model runs, suggested that it was inappropriate to fit the same model over the entire temperature gradient. Ecosystem function studies have long recognized the varying effects of temperature as a 'modulator' of ecosystem processes with various effects (Hooper, Chapin et al. 2005). With regard to the relationship between NPP and BD, temperature plays a dual role. In all cases, it is an antecedent of both NPP and BD that must be accounted for in determining the strength of the relationship between those two. However, it also appears to modulate both the strength and sign of the relationship between NPP and BD as well. At high temperatures, the strength of the relationship between BD and NPP is not as strong as the bivariate correlation coefficient indicates because of the anteceding effects of temperature. At low temperatures, the bivariate coefficient is an understatement of the strength of the relationship because temperature acts as a suppressing factor.

Further, at the low temperature end the data suggests that high biodiversity has a negative effect on NPP. For the mid-temperature range we found no strong relationship in our investigations. If data were available for other abiotic factors (soil water content, soil carbon) perhaps a relationship would surface. It is also possible that at middle range temperatures the relationship between the predictor variables and NPP is not monotonic and therefore exhibits a canceling effect.

In our high temperature range, we found NPP and diversity to be strongly linked. Assuming BD as independent, high biodiversity had a strong positive effect on NPP accounting for up to 26% of the variation. There were a number of factors we were unable to include in the model, like soil water and soil nitrogen content. These characteristics in natural systems can have large impacts on NPP and BD (Huston and McBride 2002). Since these factors are likely to interact in complex ways with the biotic and abiotic factors already included in the model it is possible that their exclusion resulted in biased estimates of model coefficients.

In this investigation we could not address causality as it is traditionally handled. The BDEF debate is particularly heated on the causality issue. On the one side the argument purports that high biodiversity drives high productivity due to more efficient resource utilization. The other side emphasizes the control of biodiversity by system productivity by mechanisms such as competition relaxation. At the same time it has been widely agreed that the relationship is bi-directional (Hooper, Chapin et al. 2005). More likely both productivity and biodiversity co-vary in a complex relationship with other factors, such as has been shown for human management of ecosystems (Cameron 2002). While the "primary" direction of causality may be important for ecological studies, it may also be impossible to discover. In addition, from a systems point of view it is not particularly relevant to talk about a "primary" direction of causality. In spite of this, the relationship between productivity and diversity has large implications for economic, ecological and policy decisions.

Ecosystem Service Value and Biodiversity

We hope that this analysis aids in understanding the complex relationships between biodiversity and ecosystem functioning. Ecosystem functioning supports ecosystem services, which are those functions of ecosystems that support human welfare, either directly or indirectly. Ecosystem services have been estimated to contribute roughly \$33 trillion/yr[1] globally to human welfare (Costanza, dArge et al. 1997). While NPP does not pick up all ecosystem services, it is a key indicator of ecosystem functioning and has been shown to correlate with the overall value of ecosystem services ((Costanza, d'Arge et al. 1998), Figure 6). This is to be expected, since NPP is a measure of the solar energy captured by the system and available to drive the functioning of the system.

In our analysis we find a strong positive relationship between biodiversity and NPP in certain temperature regimes, such that a change in biodiversity correlates with a change in NPP.

[Insert Figure 6]

We find this relationship to be dynamic at various levels of temperature (scale 2). The most compelling finding, in relation to the global loss of species, is the strong positive relationship between biodiversity and NPP at the ecoregion scale at higher temperatures. In order to assess the impact of changing diversity on the production of ecosystem services, we performed a new regression in this high temperature range using the log of NPP as the dependent variable in order to measure elasticity of NPP with respect to biodiversity. The regression equation for this was:

[1] This number was in 1994 \$US. Converting to 2004 \$US using the US Consumer Price Index yields a value of \$42 Trillion. This only adjusts for inflation, not the increasing scarcity of ecosystem services.

$\ln(NPP) = \alpha + \beta_1 {*} T + \beta_2 {*} \ln(P) + \beta_3 {*} \ln(BD)$.

The regression coefficient for ln(BD) was 0.173 ($R^2 = 0.61$, $p<0.0001$). We then combined this with earlier results for the relationship between NPP and the value of ecosystem services[2] by biome (Costanza, d'Arge et al. 1998). The equation for terrestrial biomes was:

$$\ln(V) = -12.057 + 2.599 \ln(NPP) \qquad R^2=.96,\ F=98.1,\ Prob>F=.002$$

where V is the annual value of ecosystem services in $US/ha/yr (note, however that this relationship is based on only 5 data points - Figure 5). Combining these two equations, one first sees that a one percent change in BD corresponds to a 0.173 percent change in NPP which in turn corresponds to a 0.45 percent change in ecosystems services. In other words, given the current complex relationship between biodiversity, net primary production and ecosystem services, we estimate (admittedly with fairly low precision) that a one percent loss in biodiversity in “warm” ecoregions could result in about a half a percent reduction in the value of ecosystems services provided by those regions. Another way of saying this is that the elasticity of supply of ecosystem services with respect to biodiversity is approximately 0.45.

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[2] This value was estimated from the aggregation of 17 services for 16 different biomes. Thus, a change in "value" can mean different things in different places (e.g. waste recycling verses recreational or cultural benefits). Also, while the value was estimated in dollars, it includes the full spectrum of benefits of (mainly non-marketed) ecosystem services, ranging from raw food to cultural aesthetic, and scientific benefits

On a related topic, the correlation between NPP and latitude is well known (Lieth 1978). It has been estimated that approximately 70% of the global NPP occurs in Africa and South America (Imhoff, Bounoua et al. 2004). These entire continents fall within the high temperature range of our model (average temperature 13ºC). Therefore, where the world's NPP is the highest (low latitudes), biodiversity is likely to be a crucial and positive factor. Additionally, it has been estimated that human appropriation of NPP is greater than 30% of the yearly global NPP (Vitousek, Ehrlich et al. 1986; Rojstaczer, Sterling et al. 2001). With most of global NPP occurring in low latitudes, the positive relationship between biodiversity and NPP at lower latitudes means that humanity is highly dependent on biodiversity for a large portion of its raw food, materials and other ecosystem services.

Obviously, these estimates are still fairly crude, due to biodiversity data limitations and limits on our knowledge of the links between NPP and the value of ecosystem services. As new, higher resolution data on global patterns of biodiversity, NPP, and ecosystem services become available, we will no doubt be able to significantly improve the analysis. At the same time our empirical results at two spatial scales add further texture to earlier experimental results in micro and mesocosms, and may help us to better understand the nature of the BDEF relationship across scales. We know that at larger spatial and temporal scales more biodiversity is needed to supply a steady flow of ecosystem goods and services, hence biodiversity is a key economic, social and ecological management goal (Hooper, Chapin et al. 2005). In addition to all the other reasons that biodiversity is important, it is fundamentally essential to sustain welfare of humans on the planet.

Acknowledgements

The site level analysis part of this study was a product of a problem-based course on biodiversity and ecosystem services held at the University of Vermont during the Fall semester, 2002. In addition to the authors, the following participants contributed to that analysis: Brian S. Barker, Simon C. Bird, Roelof M. J. Boumans, Marta Ceroni, Cheryl E. Frank, Erica J. Gaddis, Jennifer C. Jenkins, Michelle Johnson, Mark Keffer, Justin Kenney, Barton E. Kirk, Serguei Krivov, Caitrin E. Noel, Ferdinando Villa, Tim C. White, and Matthew Wilson. We also thank Gustavo Fonseca and Andrew Balmford for their helpful suggestions on earlier drafts of the manuscript. We also thank two additional anonymous reviewers for their helpful suggestions.

References Cited

Aarssen, L. W. (1997). "High productivity in grassland ecosystems: effected by species diversity or productive species?" Oikos **80**(1): 183-184.

Allen, T. F. H. and T. B. Starr (1982). Hierarchy: perspectives for ecological complexity. Chicago, University of Chicago Press.

Cameron, T. (2002). "2002: the year of the 'diversity-ecosystem function' debate." Trends in Ecology & Evolution **17**(11): 495-496.

Costanza, R., R. dArge, et al. (1997). "The value of the world's ecosystem services and natural capital." Nature **387**(6630): 253-260.

Costanza, R., R. d'Arge, et al. (1998). "The value of ecosystem services: putting the issues in perspective." Ecological Economics **25**(1): 67-72.

de Groot, R. S., M. A. Wilson, et al. (2002). "A typology for the classification, description and valuation of ecosystem functions, goods and services." Ecological Economics **41**(3): 393-408.

Ehrlich, P. R. and A. H. Ehrlich (1981). Extinction: The Causes and Consequences of the Disappearance of Species. New York, Ballantine Books.

Elton, C. S. (1958). Ecology of invasions by animals and plants. London, Chapman and Hall.

FAO (1995). *Digital soil map of the world and derived soil properties*. , Food and Agriculture Organization of the United Nations.

Grime, P. (1997). "Biodiversity is not an end in itself." Recherche(304): 40-41.

Heywood, V. H. (1995). Global Biodiversity Assessment. Cambridge UK, Cambridge university Press.

Hooper, D. U., F. S. Chapin, et al. (2005). "Effects of biodiversity on ecosystem functioning: A consensus of current knowledge." Ecological Monographs **75**(1): 3-35.

Huston, M. A. (1997). "Hidden treatments in ecological experiments: Re-evaluating the ecosystem function of biodiversity." Oecologia **110**(4): 449-460.

Huston, M. A. and A. C. McBride (2002). Evaluating the relative strengths of biotic versus abiotic controls on ecosystem processes. Biodiversity and ecosystem functioning: synthesis and perspectives. M. Loreau, S. Naeem and P. Inchausti. Oxford; New York, Oxford University Press**:** xii, 294.

Imhoff, M. L., L. Bounoua, et al. (2004). "Global patterns in human consumption of net primary production." Nature **429**(6994): 870-873.

Kinzig, A. P. , S.W Pacala, et. al. (2002). Functional consequences of biodiversity: empirical progress and theoretical extensions. Princeton, Princeton University Press.

Knapp, A. K. and M. D. Smith (2001). "Variation among biomes in temporal dynamics of aboveground primary production." Science **291**(5503): 481-484.

Lawton, J. H. (1998). "Pigeons, peregrines and people." Oikos **83**(2): 209-211.

Leemans, R. and W. Cramer (1991). The IIASA database for mean monthly values of temperature, precipitation and cloudiness on a global terrestrial grid. Laxenburg, Austria, International Institute of Applied Systems Analyses.

Lieth, H. F. H. (1978). Primary patterns of production in the biosphere. New York, Academic Press.

Limburg, K. E., R. V. O'Neill, et al. (2002). "Complex systems and valuation." Ecological Economics **41**(3): 409-420.

MacArthur, R. H. (1955). "Fluctuations of animal populations and a measure of community stability." Ecology(36): 533-536.

May, R. (1972). "Will a large complex system be stable?" Nature **238**: 413-414.

McCarthy, J. J. and Intergovernmental Panel on Climate Change. Working Group II. (2001). Climate change 2001: impacts, adaptation, and vulnerability: contribution of Working Group II to the third assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK; New York, Published for the Intergovernmental Panel on Climate Change [by] Cambridge University Press.

Naeem, S., K. Hakansson, et al. (1996). "Biodiversity and plant productivity in a model assemblage of plant species." Oikos **76**(2): 259-264.

Naeem, S., L. J. Thompson, et al. (1994). "Declining Biodiversity Can Alter the Performance of Ecosystems." Nature **368**(6473): 734-737.

Naeem, S., L. J. Thompson, et al. (1995). "Empirical-Evidence That Declining Species-Diversity May Alter the Performance of Terrestrial Ecosystems." Philosophical Transactions of the Royal Society of London Series B-Biological Sciences **347**(1321): 249-262.

Odum, E. P. (1953). Fundamentals of Ecology. Philadelphia, Saunders.

Paine, R. T. (2002). "Trophic controt of production in a rocky intertidal community." Science **296**(5568): 736-739.

Pfisterer, A. B. and B. Schmid (2002). "Diversity-dependent production can decrease the stability of ecosystem functioning." Nature **416**(6876): 84-86.

Ricketts, T., E. G. Dinerstein et al. (1999). Terrestrial Ecoregions of North America. Washingto DC, Island Press.

Rojstaczer, S., S. M. Sterling, et al. (2001). "Human appropriation of photosynthesis products." Science **294**(5551): 2549-2552.

Running, S. W. and J. C. Coughlan (1988). "A General-Model of Forest Ecosystem Processes for Regional Applications.1. Hydrologic Balance, Canopy Gas-Exchange and Primary Production Processes." Ecological Modelling **42**(2): 125-154.

Schulze, A. P. and M. H.A. (1993). Ecosystem Function and Biodiversity. Berlin, Springer.

Symstad, a. J., F. S. Chapin, et al. (2003). "Long-term and large-scale perspectives on the relationship between biodiversity and ecosystem functioning." Bioscience **53**(1): 89-98.

Tilman, D. and J. a. Downing (1994). "Biodiversity and Stability in Grasslands." Nature **367**(6461): 363-365.

Tilman, D., J. Knops, et al. (1997). "The influence of functional diversity and composition on ecosystem processes." Science **277**(5330): 1300-1302.

Tilman, D. and C. Lehman (2002). Biodiversity, composition and ecosystem process: theory and concepts. Functional consequences of biodiversity: empirical progress and theoretical extensions. A. P. Kinzig, S. W. Pacala and D. Tilman. Princeton, Princeton Unversity Press**:** 9-41.

Tilman, D., D. Wedin, et al. (1996). "Productivity and sustainability influenced by biodiversity in grassland ecosystems." Nature **379**(6567): 718-720.

Turner, D. P., W. D. Ritts, et al. (2003). "Scaling Gross Primary Production (GPP) over boreal and deciduous forest landscapes in support of MODIS GPP product validation." Remote Sensing of Environment **88**(3): 256-270.

Vitousek, P. M., P. R. Ehrlich, et al. (1986). "Human Appropriation of the Products of Photosynthesis." Bioscience **36**(6): 368-373.

Walker, B. H. and W. E. Steffen (1996). Global Change and Terrestrial Ecosystems. Cambridge UK, Cambridge University Press.

Wardle, D. A. and O. Zackrisson (2005). "Effects of species and functional group loss on island ecosystem properties." Nature **435**(7043): 806-810.

Wardle, D. A., O. Zackrisson, et al. (1997). "Biodiversity and ecosystem properties - Response." Science **278**(5345): 1867-1869.

Willms, W. D., J. F. Dormaar, et al. (2002). "Response of the mixed prairie to protection from grazing." Journal of Range Management **55**(3): 210-216.

Figure Legend

Figure 1. Possible causal chains between BD, NPP and abiotic factors.

Figure 2. Marginal change in NPP with biodiversity over all temperatures.

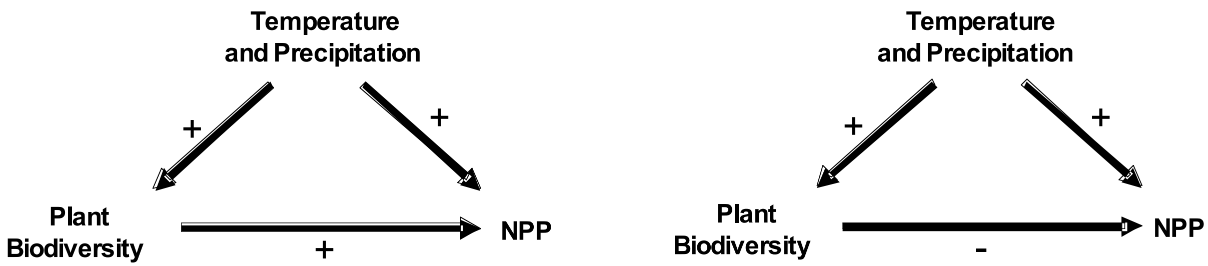
Figure 3. Scale 2 regression results over moving window regression.

Figure 4. Marginal change in NPP with biodiversity in the low temperature model.

Figure 5. Marginal change in NPP with biodiversity in the high temperature model.

Figure 6. Relationship between Net Primary Production and the value of ecosystem services by biome (from Costanza, d'Arge, et al. 1998).

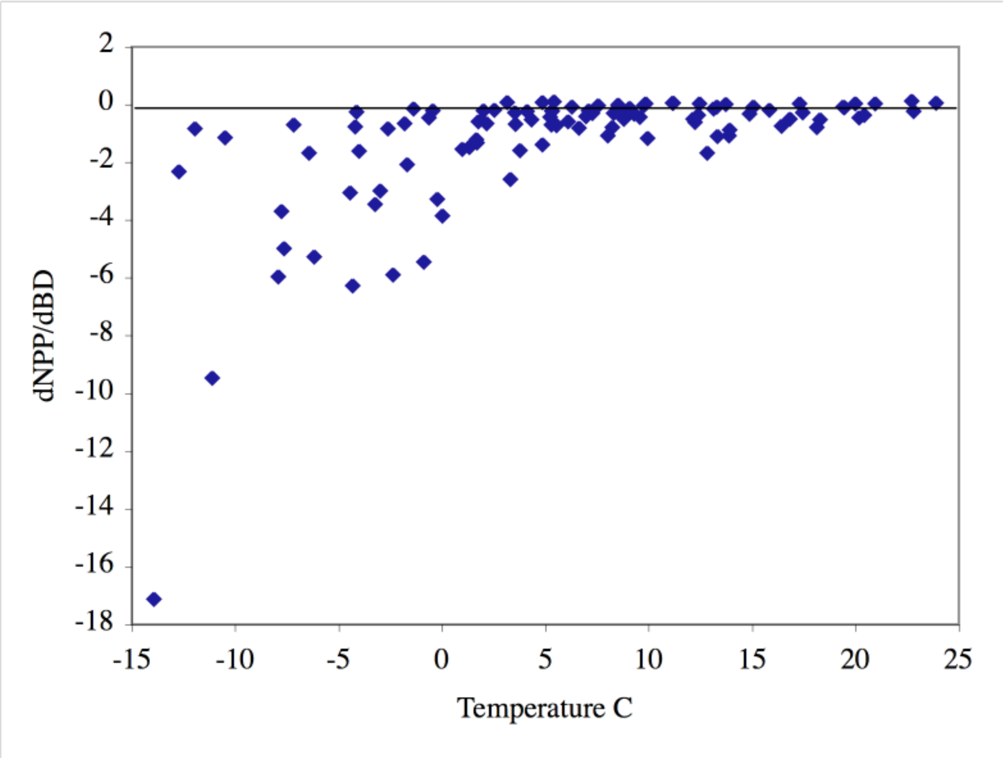
Figure 1



Case 1 – Partial Explanation

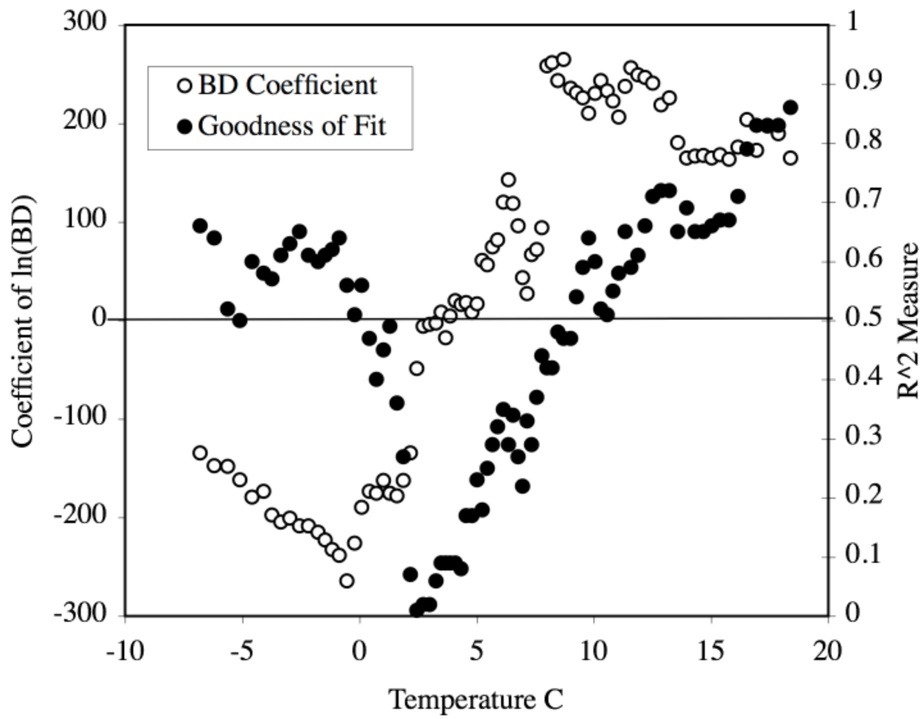
Case 2 - Suppression

Figure 2



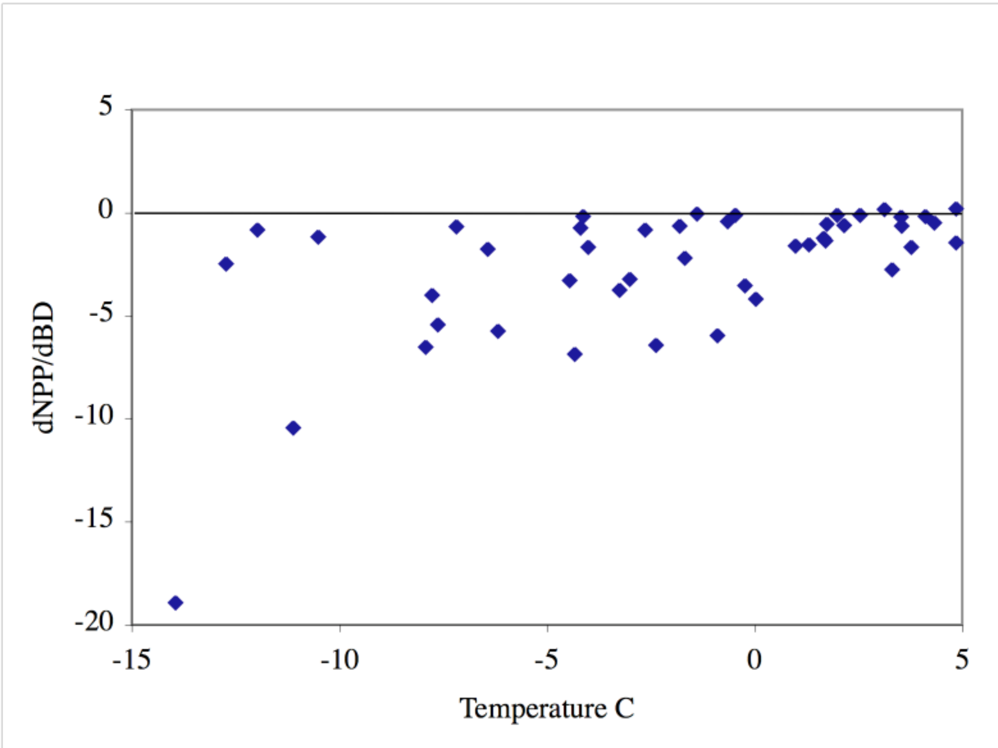
2
0
-2
-4
-6
-8
-10
-12
-14
-16
-18
dNPP/dBD
-15
-10
-5
0
5
10
15
20
25
Temperature C

Figure 3



BD Coefficient
Goodness of Fit
Coefficient of ln(BD)
R^2 Measure
Temperature C
300
200
100
0
-100
-200
-300
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
-10
-5
0
5
10
15
20

Figure 4



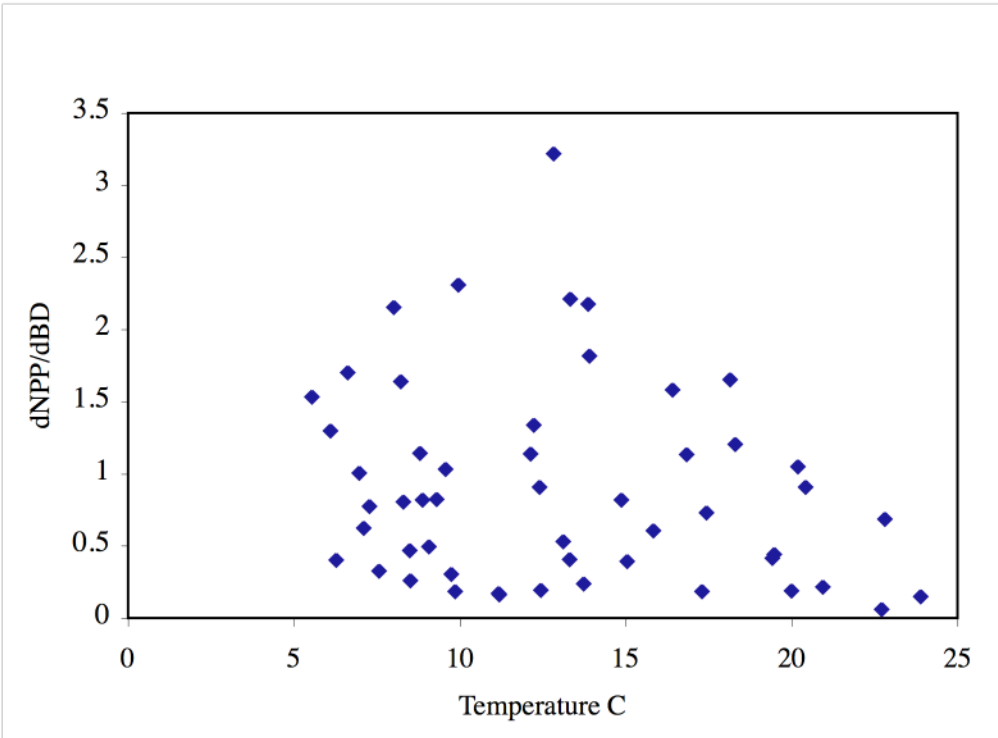


Figure 5

Figure 6

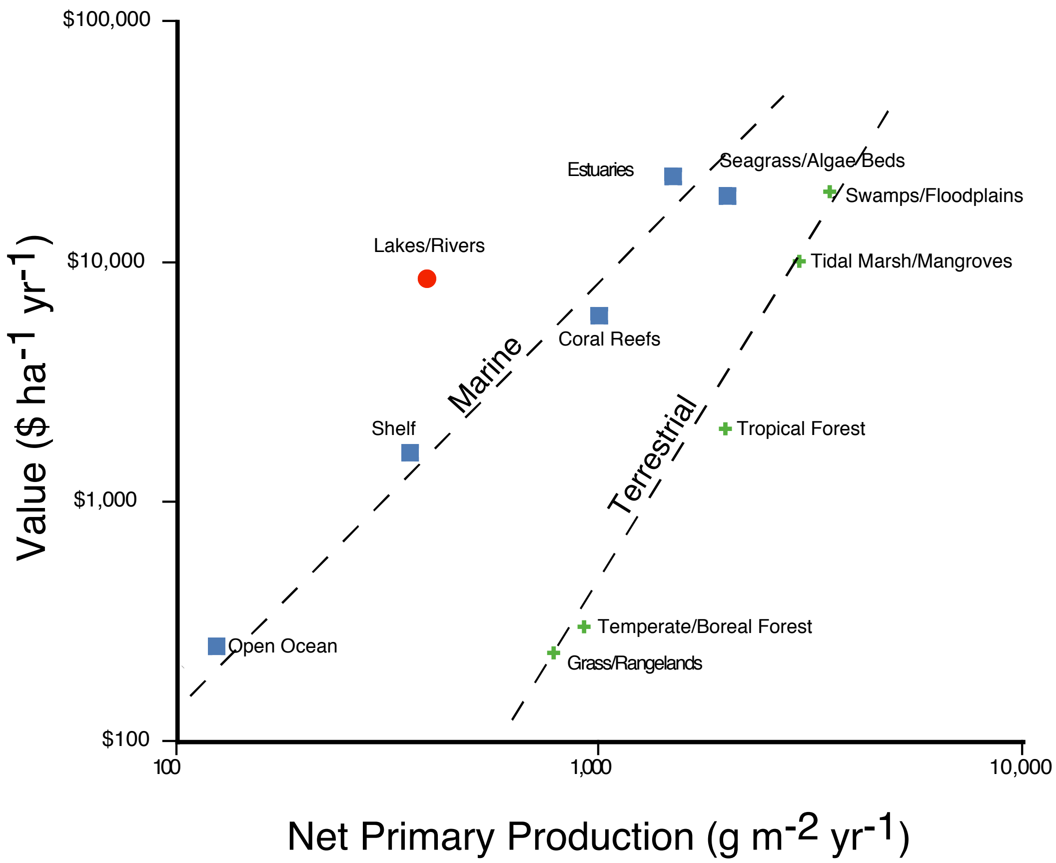

$100,000
$10,000
$1,000
$100
Value ($ ha-1 yr-1)
100
1,000
10,000
Net Primary Production (g m-2 yr-1)
Estuaries
Seagrass/Algae Beds
Swamps/Floodplains
Lakes/Rivers
Tidal Marsh/Mangroves
Coral Reefs
Marine
Shelf
Terrestrial
Tropical Forest
Temperate/Boreal Forest
Open Ocean
Grass/Rangelands

Table 1. Data used in Scale 1 (Site) NPP regression model.

Site Location	NPP (g/m2/yr)	Vascular Plants (number)	Growing Season Precipitation (mm)	Organic Carbon Upper Soil (%)	Organic Carbon Lower Soil (%)	Growing Season Temperature (Celsius)
	NPP	**BD**	**P*g***	**O*u***	**O*L***	**T**
Arctic LTER	140.75833	395	53	0.31	0.2	6.3
Bonanza Creek LTER	299.8475	214	136	2.59	0.55	11.3
Cedar Creek LTER	277.26588	796	315	0.29	0.23	20.2
Harvard Forest	744.5	225	493	3	1	20
Hubbard Brook LTER	704.5	256	482	0.44	0.28	18
Jornada LTER	229.07333	354	128	0.4	0.25	21.4
Kellogg Biological Station	430.997	436	435	0.57	0.28	19
Konza Prairie LTER	442.6	576	565	1.53	0.695	22.8
Niwot Ridge LTER	198.74267	716	108	3.2	0.94	19.4
Sevilleta LTER	184.5	822	91	0.4	0.25	20.5
Shortgrass Steppe LTER	116.5	333	217	1.83	0.87	16.4
Superior National Forest	507.65	1460	295	0.44	0.28	17.2

Table 2. Plot scale regression coefficients.

Parameter	Coefficient	Std. Error	p-value
Constant	2977.3	896.3	0.0105
Ln(BD)	-542.9	168.1	0.012
BD	0.857	0.276	0.0146
P	0.876	0.163	0.0007

Table 3. Regression coefficients for model covering entire ecoregion temperature range.

Parameter	Coefficient	Std. Error	p-value
Constant	-43.3	147.4	0.77
Ln(BD)	-103.7	46.5	0.0281
BD	0.159	0.047	0.0011
T	13.6	2	<0.0001
ln(P)	195.3	45.6	<0.0001

Table 4. Regression coefficients for low temperature ecoregions.

Parameter	Coefficient	Std. Error	p-value
Constant	78.5	81.3	0.34
ln(BD)	-115.3	43.5	0.011
BD	0.286	0.078	0.0007
ST	33.1	4.05	<0.0001

Table 5. Regression coefficients for high temperature ecoregions.

Parameter	Coefficient	Std. Error	p-value
Constant	-1011.8	172.5	<0.0001
ln(BD)	184.3	44.4	0.0001
ln(P)	333.3	54	<000.1
T	9.62	3.44	0.0075

Table S1. Data used in the ecoregion (scale 2) analysis

	Ecoregion	NPP (g/m^2/yr)	ln(NPP)	Vascular Plant Richness	Natural Area (ha)	BD (per 10,000km^2 Natural Area)	ln(BD)	Summer Temperature (Celsius)	Precipitation (mm/yr)	Average Annual Temperature (Celsius)
1	Alaska Peninsula Montane Taiga	170.69	2.23	510	3,613,116	141.15	2.15	10.54	1019	1.74
2	Alaska/St. Elias Range Tundra	98.33	1.99	747	13,147,339	56.82	1.75	8.41	838	-6.44
3	Alberta Mountain Forests	309.63	2.49	660	3,889,440	169.69	2.23	10.55	369	-0.65
4	Alberta/British Columbia Foothills Forest	529.06	2.72	740	12,026,477	61.53	1.79	14.05	420	0.98
5	Aleutian Islands Tundra	278.22	2.44	388	286,764	1353.03	3.13	8.20	925	3.13
6	Allegheny Highlands Forests	382.80	2.58	1883	8,241,231	228.49	2.36	19.93	1034	8.29
7	Appalachia/Blue Ridge Forests	572.76	2.76	2398	14,828,035	161.72	2.21	22.18	1156	12.12
8	Appalachian Mixed Mesophytic Forests	534.26	2.73	2487	18,050,094	137.78	2.14	22.43	1167	12.22
9	Arctic Coastal Tundra	90.73	1.96	539	5,107,118	105.54	2.02	5.89	111	-11.97
10	Arctic Foothills Tundra	96.93	1.99	580	7,195,035	80.61	1.91	5.94	112	-10.51
11	Arizona Mountains Forests	392.06	2.59	2204	10,854,545	203.05	2.31	22.56	151	12.39
12	Atlantic Coastal Pine Barrens	649.83	2.81	632	672,167	940.24	2.97	22.90	1058	12.43
13	Beringia Lowland Tundra	140.60	2.15	553	11,800,737	46.86	1.67	10.62	598	-1.69
14	Beringia Upland Tundra	107.61	2.03	538	9,080,866	59.25	1.77	9.65	442	-4.01
15	Blue Mountain Forests	374.65	2.57	1134	6,189,344	183.22	2.26	16.97	305	6.96
16	Brooks/British Range Tundra	95.48	1.98	593	14,158,680	41.88	1.62	5.27	150	-12.74
17	California Central Valley Grasslands	534.61	2.73	1682	3,597,998	467.48	2.67	22.03	364	15.03
18	California Coastal Sage and Chaparral	471.86	2.67	1491	1,952,235	763.74	2.88	18.10	203	13.72
19	California Interior Chaparral and Woodlan	689.42	2.84	2105	6,093,221	345.47	2.54	18.53	410	13.11
20	California Montane Chaparral and Woodland	528.68	2.72	2075	1,957,412	1060.07	3.03	16.43	255	11.15
21	Canadian Aspen Forest and Parklands	380.05	2.58	1464	22,932,526	63.84	1.81	15.91	417	1.30
22	Cascade Mountains Leeward Forests	382.98	2.58	1328	4,543,093	292.31	2.47	11.10	617	1.98
23	Central and Southern Cascades Forests	615.36	2.79	1296	4,384,978	295.55	2.47	14.71	654	7.08
24	Central and Southern Mixed Grasslands	472.33	2.67	2081	20,517,248	101.43	2.01	26.00	642	13.89
25	Central Canadian Shield Forests	481.21	2.68	1246	41,134,273	30.29	1.48	14.86	751	-0.24
26	Central Forest/Grassland Transitional Zone	529.50	2.72	2124	25,513,367	83.25	1.92	24.98	916	13.32
27	Central Pacific Coastal Forests	682.04	2.83	1109	6,878,342	161.23	2.21	13.85	1512	8.79
28	Central Tall Grasslands	356.66	2.55	1779	2,546,902	698.50	2.84	22.19	739	8.51
29	Central US Hardwood Forests	458.00	2.66	2332	27,562,726	84.61	1.93	24.52	1187	13.87
30	Chihuahuan Desert	289.92	2.46	2263	20,294,885	111.51	2.05	26.23	275	18.13
31	Colorado Plateau Shrublands	245.22	2.39	2556	32,050,685	79.75	1.90	21.22	218	9.94
32	Colorado Rockies Forests	476.41	2.68	1626	13,141,409	123.73	2.09	16.40	245	5.28
33	Cook Inlet Taiga	171.68	2.23	738	2,467,411	299.10	2.48	12.07	438	-0.47
34	Copper Plateau Taiga	161.59	2.21	407	1,549,253	262.71	2.42	9.36	973	-4.13
35	East Central Texas Forests	615.06	2.79	1553	1,593,082	974.84	2.99	28.56	940	19.99
36	Eastern Canadian Forests	404.46	2.61	1140	43,933,120	25.95	1.41	12.78	1010	0.02
37	Eastern Canadian Shield Taiga	239.35	2.38	925	57,244,775	16.16	1.21	9.95	589	-4.33
38	Eastern Cascades Forests	468.22	2.67	1224	5,169,011	236.80	2.37	16.33	393	7.27
39	Eastern Forest/ Boreal Transition	431.85	2.64	1228	32,265,635	38.06	1.58	16.97	952	3.30
40	Eastern Great Lakes Lowland Forests	311.03	2.49	1381	9,750,002	141.64	2.15	18.89	966	6.09
41	Edwards Plateau Savannas	627.34	2.80	2361	5,698,855	414.29	2.62	28.19	655	19.46
42	Everglades	942.56	2.97	1362	1,100,109	1238.06	3.09	27.73	1433	23.88
43	Flint Hills Grasslands	544.17	2.74	1174	2,607,547	450.23	2.65	25.68	842	13.29
44	Florida Sand Pine Scrub	872.04	2.94	951	311,631	3051.68	3.48	27.30	1359	22.70
45	Fraser Plateau and Basin Complex	383.67	2.58	1012	13,163,580	76.88	1.89	12.09	647	1.66
46	Great Basin Montane Forests	240.90	2.38	1043	569,664	1830.90	3.26	15.45	149	5.40
47	Great Basin Shrub Steppe	208.99	2.32	2519	29,462,050	85.50	1.93	18.58	211	8.00
48	Gulf of St. Lawrence Lowland Forests	326.09	2.51	1033	3,507,432	294.52	2.47	16.80	1300	5.32
49	Interior Alaska/Tukon Lowland Taiga	196.05	2.29	810	42,301,085	19.15	1.28	10.67	370	-6.19
50	Interior Yukon/Alaska Alpine Tundra	212.78	2.33	617	22,834,531	27.02	1.43	9.64	703	-7.78
51	Klamath-Siskiyou Forests	610.00	2.79	1859	4,739,896	392.20	2.59	14.71	554	8.48
52	Low Arctic Tundra	132.21	2.12	497	46,077,817	10.79	1.03	7.28	239	-11.12
53	Madrean Sky Islands Montane Forests	355.40	2.55	1139	1,140,862	998.37	3.00	26.73	156	17.29
54	Middle Arctic Tundra	56.47	1.75	371	61,755,681	6.01	0.78	4.09	181	-13.95
55	Middle Atlantic Coastal Forests	697.72	2.84	1488	9,165,263	162.35	2.21	25.84	1184	16.82
56	Midwestern Canadian Shield Forests	509.51	2.71	797	46,352,489	17.19	1.24	14.50	451	-2.38
57	Mississippi Lowland Forests	526.90	2.72	1468	5,846,978	251.07	2.40	26.77	1357	17.43
58	Mojave Desert	135.21	2.13	2490	11,081,656	224.70	2.35	24.58	164	14.86
59	Montana Valley and Foothill Grasslands	268.73	2.43	1197	6,742,422	177.53	2.25	16.81	325	5.22
60	Muskwa/Slave Lake Forests	507.62	2.71	722	25,100,768	28.76	1.46	13.98	342	-3.26
61	Nebraska Sandhills Mixed Grasslands	342.07	2.53	1185	5,271,180	224.81	2.35	22.30	459	8.87
62	New Engalnd/Acadian Forests	339.61	2.53	1496	22,270,268	67.17	1.83	16.55	1270	4.84
63	Newfoundland Highland Forests	410.64	2.61	473	1,542,584	306.63	2.49	12.54	1352	2.53
64	North Central Rockies Forests	358.93	2.56	1695	23,805,001	71.20	1.85	12.33	368	1.70
65	Northeastern Coastal Forests	411.69	2.61	1695	7,584,866	223.47	2.35	20.50	1114	9.29
66	Northern British Columbia Mountain Forest	292.56	2.47	909	7,056,476	128.82	2.11	10.18	519	-1.81
67	Northern California Coastal Forests	874.84	2.94	1212	1,214,663	997.81	3.00	13.25	709	9.85
68	Northern Cordillera Forests	214.61	2.33	823	25,383,183	32.42	1.51	9.97	410	-4.46
69	Northern Mixed Grasslands	270.46	2.43	1595	10,328,619	154.43	2.19	18.75	429	4.32
70	Northern Pacific Central Forests	173.64	2.24	615	4,682,783	131.33	2.12	9.95	1535	2.16
71	Northern Tall Grasslands	289.20	2.46	1055	4,236,236	249.04	2.40	19.03	497	3.53
72	Northern Transitional Alpine Forests	141.14	2.15	876	2,499,187	350.51	2.54	9.16	1018	-1.39
73	Northwest Territories Taiga	262.95	2.42	576	28,534,671	20.19	1.31	11.83	233	-7.64
74	Okanogan Forests	451.18	2.65	1355	5,074,620	267.02	2.43	14.09	419	4.11
75	Ozark Mountain Forests	673.96	2.83	1743	5,738,142	303.76	2.48	26.06	1207	15.83
76	Pacific Coastal Mountain Icefields and Tu	76.11	1.88	792	7,447,346	106.35	2.03	8.03	1273	-2.64
77	Palouse Grasslands	271.31	2.43	1290	3,465,190	372.27	2.57	19.06	422	9.06
78	Piney Woods Forests	699.00	2.84	1729	11,304,749	152.94	2.18	27.24	1274	18.29
79	Puget Sound Lowland Forests	599.69	2.78	1100	1,837,128	598.76	2.78	15.85	1025	9.73
80	Queen Charlotte Islands	383.72	2.58	459	819,493	560.10	2.75	12.20	1812	7.55
81	Sierra Nevada Forests	346.83	2.54	2373	5,200,739	456.28	2.66	13.83	233	6.26
82	Snake/Columbia Shrub Steppe	220.20	2.34	2169	19,308,886	112.33	2.05	18.61	305	8.21
83	Sonoran Desert	150.59	2.18	2068	10,219,109	202.37	2.31	28.34	188	20.40
84	South Avalon-Burin Oceanic Barrens	660.97	2.82	258	176,648	1460.53	3.16	13.05	1518	4.85
85	South Central Rockies Forests	336.65	2.53	1933	15,233,309	126.89	2.10	15.08	224	3.53
86	Southeastern Conifer Forests	787.78	2.90	3095	17,675,006	175.11	2.24	27.12	1396	20.17
87	Southeastern Mixed Forests	587.54	2.77	3363	28,871,384	116.48	2.07	25.70	1249	16.40
88	Southern Great Lakes Forests	353.89	2.55	2243	12,586,073	178.21	2.25	21.31	898	9.55
89	Southern Hudson Bay Taiga	464.57	2.67	1178	35,656,983	33.04	1.52	12.90	634	-3.01
90	Tamaulipan Mezquital	536.64	2.73	1487	5,559,790	267.46	2.43	29.69	599	22.81
91	Texas Blackland Prairies	588.33	2.77	1531	3,460,244	442.45	2.65	28.60	913	19.40
92	Torngat Mountain Tundra	92.38	1.97	286	2,323,213	123.11	2.09	4.79	480	-7.18
93	Upper Midwest Forest/ Savanna Transition	324.75	2.51	1420	13,131,875	108.13	2.03	20.25	762	6.61
94	Wasatch and Uinta Montane Forests	275.34	2.44	1109	3,953,948	280.48	2.45	16.84	222	5.30
95	Western Canadian Forests	562.97	2.75	613	33,046,364	18.55	1.27	14.96	397	-0.90
96	Western Canadian Shield Taiga	275.28	2.44	720	42,459,611	16.96	1.23	10.87	284	-7.93
97	Western Great Lakes Forests	521.06	2.72	1459	24,320,875	59.99	1.78	17.73	705	3.76
98	Western Gulf Coastal Grasslands	683.00	2.83	2165	2,560,269	845.61	2.93	28.26	1137	20.94
99	Western Short Grasslands	354.65	2.55	2359	41,245,593	57.19	1.76	24.17	444	12.82
100	Willamette Valley Forests	703.35	2.85	1067	937,610	1138.00	3.06	17.92	970	11.18
101	Wyoming Basin Shrub Steppe	183.18	2.26	1557	12,979,396	119.96	2.08	17.76	273	5.53
102	Yukon Interior Dry Forests	268.63	2.43	692	6,075,359	113.90	2.06	10.98	592	-4.19

REFERENCES

Aarssen, L. W. (1997). "High productivity in grassland ecosystems: effected by species diversity or productive species?" Oikos **80**(1): 183-184.

Ableson, P. (1979). Cost Benefit Analysis and Environmental Problems. London, Saxon House.

Adamowicz, W., J. Louviere, et al. (1994). "Combining Revealed and Stated Preference Methods for Valuing Environmental Amenities." Journal of Environmental Economics and Management **26**(3): 271-292.

Adamowicz, W. L. (2004). "What's it worth? An examination of historical trends and future directions in environmental valuation." Australian Journal of Agricultural and Resource Economics **48**(3): 419-443.

Agnello, R. J. (1989). "The Economic Value Of Fishing Success - An Application Of Socioeconomic Survey Data." Fishery Bulletin **87**(1): 223-232.

Allen, T. F. H. and T. B. Starr (1982). Hierarchy: perspectives for ecological complexity. Chcago, University of Chicago Press.

Alston, J. M., G. W. Norton, et al. (1998). Science Under Scarcity: Principles and Practice for Agricultural Research Evaluation and Priority Setting. Wallingford, UK, CAB International.

Altieri, M. A. (1991). "How Best Can We Use Biodiversity in Agroecosystems." Outlook on Agriculture **20**(1): 15-23.

Anderson, G. D. and R. C. Bishop (1986). The Valuation Problem. Natural resource economics: policy problems and contemporary analysis. D. W. Bromley. Boston, MA, Liuwer Nijoff Publishing: 89-137.

Anderson, G. D. and S. F. Edwards (1986). "Protecting Rhode-Island Coastal Salt Ponds - an Economic-Assessment of Downzoning." Coastal Zone Management Journal **14**(1-2): 67-91.

Anderson, J. E. (1976). "The social cost of input distortions: a comment and a generalization." American Economic Review **66**(1): 235-238.

Andow, D. A. (1991). "Vegetational Diversity and Arthropod Population Response." Annual Review of Entomology **36**: 561-586.

Anielski, A. and S. Wilson (2005). Counting Canada's Natural Capital: Assessing the Real Value of Canada's Boreal Ecosystems, The Pembina Institute.

Anielski, M. and S. Wilson (2005). The real wealth of the Mackenzie Region: assessing the natural capital values of a northern boreal ecosystem, Canadian Boreal Initiative.

Argady, T., J. Alder, et al. (2005). Coastal Systems: An Assessment of Ecosystem Services. Ecosystems and Human Well-Being: Current States and Trends (Millennium Ecosystem Assessment). D. C. Washington, Island Press: 515-549.

Arin, T. and R. A. Kramer (2002). "Divers' willingness to pay to visit marine sanctuaries: an exploratory study." Ocean & Coastal Management **45**(2-3): 171-183.

Armbrecht, I. and I. Perfecto (2003). "Litter-twig dwelling ant species richness and predation potential within a forest fragment and neighboring coffee plantations of contrasting habitat quality in Mexico." Agriculture Ecosystems & Environment **97**(1-3): 107-115.

Arrow, K. and H. Raynaud (1986). Social choice and multicriterion decision-making.

Cambridge, MA, MIT Press.

Arrow, K., R. Solow, et al. (1993). Federal Register **58**: 4602-4613.

Arrow, K. J., M. L. Cropper, et al. (1996). "Is There a Role for Benefit-Cost Analysis in Environmental, Health, and Safety Regulation?" Science **272**(5259): 221-222.

Arrow, K. J. and A. C. Fisher (1974). "Environmental Preservation, Uncertainty, and Irreversibility." Quarterly Journal of Economics **88**: 312-319.

Asafu-Adjaye, J., R. Brown, et al. (2005). "On measuring wealth: a case study on the state of Queensland." Journal Of Environmental Management **75**(2): 145-155.

Azar, C. and T. Sterner (1996). "Discounting and distributional considerations in the context of global warming." Ecological Economics **19**(2): 169-184.

Balmford, A., A. Bruner, et al. (2002). "Ecology - Economic reasons for conserving wild nature." Science **297**(5583): 950-953.

Barbier, E. B. (2000). "Valuing the environment as input: review of applications to mangrove-fishery linkages." Ecological Economics **35**(1): 47-61.

Barbier, E. B. (2007). "Valuing ecosystem services as productive inputs." Economic Policy(49): 178-229.

Barbier, E. B. and I. Strand (1998). "Valuing mangrove-fishery linkages. A case study of Campeche, Mexico." Environmental and Resource Economics **12**(2): 151-166.

Barde, J.-P. and D. W. Pearce (1991). Introduction. Valuing the environment: six case studies. J.-P. Barde and D. W. Pearce. London, Earthscan Publications**:** 1-8.

Bateman, I. J., M. A. Cole, et al. (2006). "Comparing contingent valuation and contingent ranking: A case study considering the benefits of urban river water quality improvements." Journal of Environmental Management **79**(3): 221-231.

Bateman, I. J. and A. P. Jones (2003). "Contrasting conventional with multi-level modeling approaches to meta-analysis: Expectation consistency in UK woodland recreation values." Land Economics **79**(2): 235-258.

Beach, D. (2002). Coastal Sprawl: The Effects of Urban Design on Aquatic Ecosystems in the United States. Arlington, VA, Pew Oceans Commission.

Beatley, T., D. J. Brower, et al. (2002). An Introduction to Coastal Zone Management. Washington, DC, Island Press.

Bell, F. W. (1997). "The economic valuation of saltwater marsh supporting marine recreational fishing in the southeastern United States." Ecological Economics **21**(3): 243-254.

Bell, F. W. and V. R. Leeworthy (1990). "Recreational Demand by Tourists for Saltwater Beach Days." Journal of Environmental Economics and Management **18**(3): 189-205.

Bennett, J., M. Morrison, et al. (1998). "Testing the validity of responses to contingent valuation questioning." Australian Journal Of Agricultural And Resource Economics **42**(2): 131-148.

Benton, T. G., J. A. Vickery, et al. (2003). "Farmland biodiversity: is habitat heterogeneity the key?" Trends in Ecology & Evolution **18**(4): 182-188.

Bergstrom, J. C. and P. De Civita (1999). "Status of benefits transfer in the United States and Canada: A review." Canadian Journal of Agricultural Economics-Revue Canadienne d'Agroeconomie **47**(1): 79-87.

Bergstrom, J. C., J. H. Dorfman, et al. (2004). "Estuary management and recreational fishing benefits." Coastal Management **32**(4): 417-432.

Bergstrom, J. C., J. R. Stoll, et al. (1990). "Economic value of wetlands-based recreation." Ecological Economics **2**(2): 129-147.

Bergstrom, J. C. and L. O. Taylor (2006). "Using meta-analysis for benefits transfer: Theory and practice." Ecological Economics **60**(2): 351-360.

Bhat, M. G. (2003). "Application of non-market valuation to the Florida Keys marine reserve management." Journal of Environmental Management **67**(4): 315-325.

Bingham, G., R. Bishop, et al. (1995). "Issues in Ecosystem Valuation - Improving Information for Decision-Making." Ecological Economics **14**(2): 73-90.

Bishop, R. (1993). "Economic efficiency, sustainability, and biodiversity." Ambio **22**: 69-73.

Bishop, R. C., P. A. Champ, et al. (1997). Measuring Non-Use Values: Theory and Empirical Applications. Determining the Value of Non-Marketed Goods. R. G. Kopp, W. W. Pommerehne and N. Schwarz. Boston, Kluwer Academic Publishers**:** 59-81.

Bockstael, N., R. Costanza, et al. (1995). "Ecological Economic Modeling and Valuation of Ecosystems." Ecological Economics **14**(2): 143-159.

Bockstael, N. E., A. M. Freeman, et al. (2000). "On measuring economic values for nature." Environmental Science & Technology **34**(8): 1384-1389.

Bockstael, N. E., K. E. McConnell, et al. (1989). "A random utlity model for sportfishing: some preliminary results for Florida." Marine Resource Economics **6**: 245-260.

Bockstael, N. E., K. E. McConnell, et al. (1989). "Measuring the Benefits of Improvements in Water Quality: The Chesapeake Bay." Marine Resource Economics **6**: 1-18.

Bommarco, R. (1998). "Reproduction and energy reserves of a predatory carabid beetle relative to agroecosystem complexity." Ecological Applications **8**(3): 846-853.

Bommarco, R. and J. E. Banks (2003). "Scale as modifier in vegetation diversity experiments: effects on herbivores and predators." Oikos **102**(2): 440-448.

Bonnieux, F. and P. Rainelli (1999). Contingent valuation methodology and the EU institutional framework. Valuing environmental preferences. I. Bateman and M. Williams. New York, Oxford University Press: 585-613.

Boulding, K. (1966). The Economics of the Coming Spaceship Earth. Environmental Quality in a Growing Economy. H. Jarrett. Baltimore, John Hopkins Press.

Boumans, R., R. Costanza, et al. (2002). "Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model." Ecological Economics **41**(3): 529-560.

Boxall, P. C., W. L. Adamowicz, et al. (1996). "A comparison of stated preference methods for environmental valuation." Ecological Economics **18**(3): 243-253.

Boyd, J. and S. Banzhaf (2006). What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units. RFF DP 06-02. Washington, D.C., Resources for the future.

Boyle, K. J., G. L. Poe, et al. (1994). "What Do We Know About Groundwater Values - Preliminary Implications from a Metaanalysis of Contingent-Valuation Studies." American Journal of Agricultural Economics **76**(5): 1055-1061.

Brander, L. M., R. Florax, et al. (2006). "The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature." Environmental & Resource Economics **33**(2): 223-250.

Breaux, A., S. Farber, et al. (1995). "Using Natural Coastal Wetlands Systems for Waste-Water Treatment - an Economic Benefit Analysis." Journal of Environmental Management **44**(3): 285-291.

Breunig, K. (2003). Losing Ground: At What Cost? Changes in Land Uses and Their Impact on Habitat, Biodiversity, and Ecosystem Services in Massachusetts Massachusetts Audubon Summary Report.

Briol, E. (2004). Valuing Agricultural Biodiversity on Home Gardens in Hungary: An Application of Stated and Revealed Preference Methods, University College London, University of London. **PhD thesis**.

Brookshire, D. S. and H. R. Neill (1992). "Benefit Transfers - Conceptual and Empirical Issues." Water Resources Research **28**(3): 651-655.

Brouwer, R. (2000). "Environmental value transfer: state of the art and future prospects." Ecological Economics **32**(1): 137-152.

Brouwer, R., I. H. Langford, et al. (1997). A meta-analysis of wetland contingent valuation studies. Norwich, CSERGE, University of East Anglia.

Brouwer, R. and F. A. Spaninks (1999). "The validity of environmental benefits transfer: Further empirical testing." Environmental & Resource Economics **14**(1): 95-117.

Bullock, J. M., R. F. Pywell, et al. (2001). "Restoration of biodiversity enhances agricultural production." Ecology Letters **4**(3): 185-189.

Burd, M. (1994). "Bateman Principle and Plant Reproduction - the Role of Pollen Limitation in Fruit and Seed Set." Botanical Review **60**(1): 83-139.

Cameron, T. A. (1992). "Combining Contingent Valuation and Travel Cost Data for the Valuation of Nonmarket Goods." Land Economics **68**(3): 302-317.

Cameron, T. A. (2002). "2002: the year of the 'diversity-ecosystem function' debate." Trends in Ecology & Evolution **17**(11): 495-496.

Cameron, T. A. and D. D. Huppert (1989). "Ols Versus Ml Estimation of Non-Market Resource Values with Payment Card Interval Data." Journal of Environmental Economics and Management **17**(3): 230-246.

Cameron, T. A. and M. D. James (1987). "Efficient Estimation Methods for Closed-Ended Contingent Valuation Surveys." Review of Economics and Statistics **69**(2): 269-276.

Campell, B. and M. Luckert, Eds. (2002). Uncovering the Hidden Harvest: Valuation Methods or Woodland and Forest Resources. , . London, Earthscan.

Cane, J. H. and V. J. Tepedino (2001). "Causes and extent of declines among native North American invertebrate pollinators: Detection, evidence, and consequences." Conservation Ecology **5**(1): art. no.-1.

Cantrell, R. N., M. Garcia, et al. (2004). "Recreational anglers' willingness to pay for increased catch rates of Pacific threadfin (Polydactylus sexfilis) in Hawaii." Fisheries Research **68**(1-3): 149-158.

Carr, L. and R. Mendelsohn (2003). "Valuing coral reefs: A travel cost analysis of the Great Barrier Reef." Ambio **32**(5): 353-357.

Carson, R. T. (2000). "Contingent valuation: A user's guide." Environmental Science & Technology **34**(8): 1413-1418.

Carson, R. T., N. E. Flores, et al. (2001). "Contingent valuation: Controversies and evidence." Environmental & Resource Economics **19**(2): 173-210.

Carson, R. T., M. Hanemann, et al. (1992). A contingent valuation study of lost passive use values resulting from the Exxon Valdez oil spill. Anchorage, Attorney General of the State of Alaska.

Carson, R. T., R. C. Mitchell, et al. (2003). "Contingent valuation and lost passive use: Damages from the Exxon Valdez oil spill." Environmental & Resource Economics **25**(3): 257-286.

Champ, P. A., K. J. Boyle, et al., Eds. (2003). A primer on nonmarket valuation. Dordrecht, Kluwer Academic Press.

Chapin, F. S., E. S. Zavaleta, et al. (2000). "Consequences of changing biodiversity." Nature **405**(6783): 234-242.

Chen, W. Q., H. S. Hong, et al. (2004). "Recreation demand and economic value: An application of travel cost method for Xiamen Island." China Economic Review **15**(4): 398-406.

Chichilnisky, G. and G. Heal (1998). "Economic returns from the biosphere - Commentary." Nature **391**(6668): 629-630.

Ciriacy-Wantrup, S. (1947). "Capital returns from soil conservation practices." Journal of Farm Economics **29**: 1181-1196.

Clawson, M. (1959). Methods of measuring the demand for and value of outdoor recreation. Washington, D.C., Resource for the Future.

Cleveland, C. J., M. Betke, et al. (2006). "Economic value of the pest control service provided by Brazilian free-tailed bats in south-central Texas." Frontiers in Ecology and the Environment **5**(5): 238-243.

Cohen, J. E., C. Small, et al. (1997). "Estimates of coastal populations." Science **278**(5341): 1211-1212.

Conway, G. R. (1987). "The Properties of Agroecosystems." Agricultural Systems **24**(2): 95-117.

Coote, A. and J. Lenaghan (1997). Citizen juries: theory into practice. London, Institute for Public Policy Research.

Costanza, R. (1980). "Embodied energy and economic valuation." Science **225**: 890-897.

Costanza, R. (2000). The Ecological, Economic and Social Importance of the Oceans. Seas at the Millenium: An Environmental Evaluation. C. R. C. Sheppard. New York, Pergamon**:** 393-403.

Costanza, R. (2000). "Social goals and the valuation of ecosystem services." Ecosystems **3**(1): 4-10.

Costanza, R. (2006). "Nature: ecosystems without commodifying them." Nature **443**(7113): 749-749.

Costanza, R. and H. E. Daly (1992). "Natural Capital and Sustainable Development." Conservation Biology **6**(1): 37-46.

Costanza, R., R. d'Arge, et al. (1998). "The value of ecosystem services: putting the issues in perspective." Ecological Economics **25**(1): 67-72.

Costanza, R., R. dArge, et al. (1998). "Auditing the earth: Costanza and his coauthors reply." Environment **40**: 26-27.

Costanza, R., R. d'Arge, et al. (1997). "The value of the world's ecosystem services and natural capital." Nature **387**(6630): 253-260.

Costanza, R., S. C. Farber, et al. (1989). "Valuation and management of wetland ecosystems." Ecological Economics **1**(4): 335-361.

Costanza, R. and C. Folke (1997). Valuing ecosystem services with efficiency, fairness and sustainability as goals. Nature's services: societal dependence on natural ecosystems. G. Daily. Washington, D.C., Island Press**:** 49-68.

Costanza, R. and R. A. Herendeen (1984). "Embodied Energy and Economic Value in the United-States-Economy - 1963, 1967 and 1972." Resources and Energy **6**(2): 129-163.

Costanza, R., A. Voinov, et al. (2002). "Integrated ecological economic modeling of the Patuxent River watershed, Maryland." Ecological Monographs **72**(2): 203-231.

Costanza, R., M. Wilson, et al. (2007). The Value of New Jersey's Ecosystem Services and Natural Capital, New Jersey Department of Environmental Protection.

Cragg, R. G. and R. D. Bardgett (2001). "How changes in soil faunal diversity and composition within a trophic group influence decomposition processes." Soil Biology and Biochemistry **33**: 2073–2081.

Crocker, T. D. (1999). A short history of environmental and resource economics. Handbook of environmental and resource economics. J. van den Bergh. Cheltenham, U.K., Edward Elgar**:** 32-45.

Daily, G. (1997). Nature's services: societal dependence on natural ecosystems. Washington, D.C., Island Press.

Daily, G. C., G. Ceballos, et al. (2003). "Countryside biogeography of neotropical mammals: Conservation opportunities in agricultural landscapes of Costa Rica." Conservation Biology **17**(6): 1814-1826.

Daly, H. E. (1977). Steady-State Economics. Washington, D.C., Island Press.

Daly, H. E. (1992). "Allocation, distribution, and scale: towards an economics that is efficient, just, and sustainable." Ecological Economics **6**: 185-193.

David Evans and Associates, I. and ECONorthwest (2004). Final submittal: comparative valuation of ecosystem services: Lents project case study.

Davis, R. K. (1963). "Recreation planning as an economic problem." Natural Resources Journal **3**: 239-249.

de Groot, R. S., M. A. Wilson, et al. (2002). "A typology for the classification, description and valuation of ecosystem functions, goods and services." Ecological Economics **41**(3): 393-408.

Desvousges, W. H., F. R. Johnson, et al. (1998). Environmental policy analysis with limited information: principles and application of the transfer method., Edward Elgar.

Desvousges, W. H., M. C. Naughton, et al. (1992). "Benefit Transfer - Conceptual Problems in Estimating Water-Quality Benefits Using Existing Studies." Water Resources Research **28**(3): 675-683.

Desvousges, W. H., V. K. Smith, et al. (1987). "Option price estimates for water quality improvements: A contingent valuation study for the Monongahela river." Journal of Environmental Economics and Management **14**(3): 248-267.

Di Falco, S. and C. Perrings (2003). "Crop genetic diversity, productivity and stability of agroecosystems. A theoretical and empirical investigation." Scottish Journal of Political Economy **50**(2): 207-216.

Diamond, P. A. and J. A. Hausman (1994). "Contingent Valuation - Is Some Number Better Than No Number." Journal of Economic Perspectives **8**(4): 45-64.

Downing, M. and T. Ozuna (1996). "Testing the reliability of the benefit function transfer approach." Journal Of Environmental Economics And Management **30**(3):

316-322.

Drake, L. (1992). "The non-market value of the Swedish agricultural landscape." European Review of Agricultural Economics **19**: 351-364.

Drucker, A. G., V. Gomez, et al. (2001). "The economic valuation of farm animal genetic resources: a survey of available methods." Ecological Economics **36**(1): 1-18.

Duffus, D. A. and P. Dearden (1993). "Recreational Use, Valuation, And Management, Of Killer Whales (Orcinus-Orca) On Canada Pacific Coast." Environmental Conservation **20**(2): 149-156.

Eade, J. D. O. and D. Moran (1996). "Spatial economic valuation: Benefits transfer using geographical information systems." Journal of Environmental Management **48**(2): 97-110.

Earnhart, D. (2001). "Combining revealed and stated preference methods to value environmental amenities at residential locations." Land Economics **77**(1): 12-29.

Edwards, S. F. and F. J. Gable (1991). "Estimating the value of beach recreation from property values: an exploration with comparisons to nourishment costs." Ocean & Shoreline Management **15**: 37-55.

Edwards-Jones, G., E. S. Edwards-Jones, et al. (1995). "A Comparison of Contingent Valuation Methodology and Ecological Assessment as Techniques for Incoporating Ecological Goods into Land-use Decisions." Journal of Environmental Planning and Management **38**(2): 215-230.

Ehrlich, P. and A. Ehrlich (1981). Extinction: the causes and consequences of the disappearces of species. New York, Random House.

Ehrlich, P., A. Ehrlich, et al. (1977). Ecoscience: population, resources, environment. .

San Francisco, W.H. Freeman.

Ehrlich, P. R. and E. O. Wilson (1991). "Biodiversity Studies - Science and Policy." Science **253**(5021): 758-762.

Elton, C. S. (1958). Ecology of Invasions by animals and plants. London, Chapman and Hall.

EPA, U. S. (2000). Guidelines for Preparing Economic Analyses. EPA 240-R-00-003. . Washington, D.C., U.S. Environmental Protection Agency.

Evenson, R. E. and D. Gollin (2003). "Assessing the impact of the Green Revolution, 1960 to 2000." Science **300**(5620): 758-762.

Eviner, V. T. and F. S. Chapin (2003). Biogeochemical interactions and biodiversity. Interactions of the Major Biogeochemical Cycles--Global Change and Human Impacts. J. M. Melillo, C. B. Fileld and B. Moldan. Washington, D.C., Island Press: 151-173.

FAO (1995). Digital soil map of the world and derived soil properties. Rome.

Farber, S. (1987). "The Value of Coastal Wetlands for Protection of Property against Hurricane Wind Damage." Journal of Environmental Economics and Management **14**(2): 143-151.

Farber, S. (1988). "The Value of Coastal Wetlands for Recreation - an Application of Travel Cost and Contingent Valuation Methodologies." Journal of Environmental Management **26**(4): 299-312.

Farber, S. and R. Costanza (1987). "The Economic Value of Wetlands Systems." Journal of Environmental Management **24**(1): 41-51.

Farber, S., R. Costanza, et al. (2006). "Linking ecology and economics for ecosystem management." Bioscience **56**(2): 121-133.

Farber, S., R. Costanza, et al. (2002). "Economic and Ecological Concepts for Valuing Ecosystem Services." Ecological Economics **41**(3): 375-392.

Farber, S. and B. Griner (2000). "Using conjoint analysis to value ecosystem change." Environmental Science & Technology **34**(8): 1407-1412.

Fleischer, A. and Y. Tsur (2000). "Measuring the recreational value of agricultural landscape." European Review of Agricultural Economics **27**(3): 385-398.

Flores, N. E. and J. Thacher (2002). "Money, who needs it? Natural resource damage assessment." Contemporary Economic Policy **20**(2): 171-178.

Folke, C., J. Colding, et al. (2003). Synthesis: Building resilience and adaptive capacity in social-ecological systems. Navigating Social-Ecological Systems: Building Resilience for Complexity and Change. F. Berkes, J. Colding and C. Folke. Cambridge, UK., Cambridge University Press**:** 352-387.

Font, A. R. (2000). "Mass tourism and the demand for protected natural areas: A travel cost approach." Journal Of Environmental Economics And Management **39**(1): 97-116.

Freckman, D. W. and R. A. Virginia (1997). "Low-diversity Antarctic soil nematode communities: Distribution and response to disturbance." Ecology **78**(2): 363-369.

Freeman III, A. K. (1993). The measurement of environmental and resources values. Washington, D,C,, Resource for the Future.

Freeman III, A. K. (2003). The measurement of environmental and resources values. Washington, DC, Resource for the Future.

French, P. W. (1997). Coastal and Estuarine Management. New York, Routledge.

Froehlich, M., D. J. C. Hufford, et al. (1991). The United States. Valuing the environment: six case studies. J.-P. Barde and D. W. Pearce. London, Earthscan Publications Limited.

Gandini, G. C. and E. Villa (2003). "Analysis of the cultural value of local livestock breeds: a methodology." Journal of Animal Breeding and Genetics **120**(1): 1-11.

Georgescu-Roegen, N. (1971). The entropy law and the economic process. Cambridge, MA, Harvard University Press.

Giraud, K., B. Turcin, et al. (2002). "Economic benefit of the protection program for the Steller sea lion." Marine Policy **26**(6): 451-458.

Gitay, H., J. B. Wilson, et al. (1996). "Species redundancy: A redundant concept?" Journal of Ecology **84**(1): 121-124.

Glass, G. V. (1976). "Primary, secondary, and meta-analysis of research." Educational Researcher **5**: 3-8.

Gosselink, J. G., E. G. Odum, et al. (1974). The value of the tidal marsh, Center for Wetland Resources, Louisiana State University.

Goulder, L. H. and D. Kennedy (1997). Valuing Ecosystem Services: Philosophical Bases and Empirical Method. Nature's Services: Societal Dependence on Natural Ecosystems. G. Daily. Washington, DC, Island Press: 23-48.

Gren, I.-M. (2003). Monetary green accounting and ecosystem services. Working papre No.86, July 2003. Stockholm, The National Institute of Economic Research.

Gret-Regamey, A. and S. Kytzia (2007). "Integrating the valuation of ecosystem services into the Input-Output economics of an Alpine region." Ecological Economics **63**(4): 786-798.

Griffiths, B. S., K. Ritz, et al. (2000). "Ecosystem response of pasture soil communities to fumigation-induced microbial diversity reductions: an examination of the biodiversity-ecosystem function relationship." Oikos **90**(2): 279-294.

Griffiths, C. (2002). "The use of benefit-cost analyses in environmental policy making." from http://www.bus.ucf.edu/mdickie/Health%20Workshop/Papers/Griffiths.pdf.

Grime, P. (1997). "Biodiversity is not an end in itself." Recherche(304): 40-41.

Hahn, R. W. (2000). "The impact of economics on environmental policy." Journal of Environmental Economics and Management **39**(3): 375-399.

Hanemann, M. (1992). Preface. Pricing the European environment. S. Navrud. Oslo, Scandinavian University Press**:** 9-35.

Hanemann, W. M. (1994). "Valuing the Environment through Contingent Valuation." Journal of Economic Perspectives **8**(4): 19-43.

Hanley, N., D. MacMillan, et al. (1998). "Contingent valuation versus choice experiments: Estimating the benefits of environmentally sensitive areas in Scotland." Journal of Agricultural Economics **49**(1): 1-15.

Hanley, N. and C. Spash (1993). Cost-Benefit Analysis and the Environment. Cornwall, Edward Elgar Publishing.

Hausman, J. A. (1993). Contingent valuation: a critical assessment. Amsterdam, North-Holland.

Hausman, J. A., G. K. Leonard, et al. (1995). "A Utility-Consistent, Combined Discrete-Choice and Count Data Model Assessing Recreational Use Losses Due to Natural-Resource Damage." Journal of Public Economics **56**(1): 1-30.

Hayes, K. M., T. J. Tyrrell, et al. (1992). "Estimating the benefits of water quality improvements in the Upper Narragansett Bay." Marine Resource Economics **7**: 75-85.

Heal, G. M., E. B. Barbier, et al. (2005). Valuing Ecosystem Services: Toward Better Environmental Decision-Making. Washington, DC, The National Academies Press.

Hector, A., B. Schmid, et al. (1999). "Plant diversity and productivity experiments in European grasslands." Science **286**(5442): 1123-1127.

Heisey, P. W., M. Smale, et al. (1997). "Wheat rusts and the cost of genetic diversity in the Punjab of Pakistan." American Journal of Agricultural Economics **79**(3): 726-737.

Heywood, V. H. (1995). Global Biodiversity Assessment. Cambridge, Cambridge University Press.

Higgins, S. I., J. K. Turpie, et al. (1997). "An ecological economic simulation model of mountain fynbos ecosystems - Dynamics, valuation and management." Ecological Economics **22**(2): 155-169.

Hinrichsen, D. (1998). Coastal Waters of The World: Trends, Threats, and Strategies. Washington, DC, Island Press.

Hoehn, J. P. (2006). "Methods to address selection effects in the meta regression and transfer of ecosystem values." Ecological Economics **60**(2): 389-398.

Holling, C. S. (1973). "Resilience and stability of ecological systems. Annual Review of Ecology and Systematics " Annual Review of Ecology and Systematics **4**: 1-23.

Holling, C. S., D. S. Schindler, et al. (1995). Biodiversity in the functioning of ecosystems: An ecological synthesis. . Biodiversity Loss: Economic and

Ecological Issues. C. Perrings, K.-G. Mäler, C. Folke, C. S. Holling and B.-O. Jansson. Cambridge, U.K., Cambridge University Press: 44-83.

Honey, M. (1999). Ecotourism and Sustainable Development: Who Owns the Paradise? . Washington, D.C., Island Press.

Hooper, D. U., F. S. Chapin, et al. (2005). "Effects of biodiversity on ecosystem functioning: A consensus of current knowledge." Ecological Monographs **75**(1): 3-35.

Hooper, D. U. and P. M. Vitousek (1997). "The effects of plant composition and diversity on ecosystem processes." Science **277**(5330): 1302-1305.

Horowitz, J. K. and K. E. McConnell (2003). "Willingness to accept, willingness to pay and the income effect." Journal of Economic Behavior & Organization **51**(4): 537-545.

Hotelling, H. (1949). Letter to the Director of National Park Service. The economics of public recreation: the Prewitt report. R. A. Prewitt. Washington, D.C., Department of Interior.

Howarth, R. B. and S. Farber (2002). "Accounting for the value of ecosystem services." Ecological Economics **41**(3): 421-429.

Howarth, R. B. and M. A. Wilson (2006). "A Theoretical Approach to Deliberative Valuation: Aggregation by Mutual Consent." Land Economics **82**: 1-16.

Huppert, D. D. (1989). "Measuring the value of fish to anglers: application to central California Anadromous species." Marine Resource Economics **6**: 89-107.

Huston, M. A. (1997). "Hidden treatments in ecological experiments: Re-evaluating the ecosystem function of biodiversity." Oecologia **110**(4): 449-460.

Huston, M. A. and A. C. McBride (2002). Evaluating the relative strengths of biotic versus abiotic controls on ecosystem processes. Biodiversity and ecosystem functioning--synthesis and perspectives. M. Loreau, S. Naeem and P. Inchausti. Oxford, Oxford University Press**:** 294.

Hwang, C. L. and K. Yoon (1981). Multiple Attribute Decision Making: Methods and Applications – A State of the Art Survey. New York, Spriner-Verlag.

Imhoff, M. L., L. Bounoua, et al. (2004). "Global patterns in human consumption of net primary production." Nature **429**(6994): 870-873.

Jacobs, M. (1997). Environmental valuation, deliberative democracy and public decision-making. Valuing Nature: Economics, Ethics and Environment,. J. Foster. London, Rutledge**:** 211-231.

Jiang, Y., S. K. Swallow, et al. (2005). "Context-sensitive benefit transfer using stated choice models: Specification and convergent validity for policy analysis." Environmental & Resource Economics **31**(4): 477-499.

Johnston, R. J., E. Y. Besedin, et al. (2005). "Systematic variation in willingness to pay for aquatic resource improvements and implications for benefit transfer: a meta-analysis." Canadian Journal of Agricultural Economics-Revue Canadienne D Agroeconomie **53**(2-3): 221-248.

Johnston, R. J., J. J. Opaluch, et al. (2001). "Estimating amenity benefits of coastal farmland." Growth and Change **32**: 305-325.

Jones, C. G., J. H. Lawton, et al. (1994). "Organisms as Ecosystem Engineers." Oikos **69**(3): 373-386.

Just, R. E., L. H. Darrell, et al. (1982). "Applied welfare economics and public policy."

American Economic Review **69**(5): 947-954.

Just, R. E. and D. L. Heuth (1979). Applied welfare economics and public policy. Englewood Cliffs, NJ, Prentice-Hall.

Kahn, J. R. and R. B. Buerger (1994). "Valuation and the Consequences of Multiple Sources of Environmental Deterioration - the Case of the New-York Striped Bass Fishery." Journal of Environmental Management **40**(3): 257-273.

Kahneman, D. and J. L. Knetsch (1992). "Valuing Public-Goods - the Purchase of Moral Satisfaction." Journal of Environmental Economics and Management **22**(1): 57-70.

Kaoru, Y. (1993). "Differentiating Use and Nonuse Values for Coastal Pond Water Quality Improvements." Environmental & Resource Economics **3**: 487-494.

Kaoru, Y., V. K. Smith, et al. (1995). "Using Random Utility-Models to Estimate the Recreational Value of Estuarine Resources." American Journal of Agricultural Economics **77**(1): 141-151.

Kawabe, M. and T. Oka (1996). "Benefit from improvement of organic contamination of Tokyo Bay." Marine Pollution Bulletin **32**(11): 788-793.

King, O. H. (1995). "Estimating the value of marine resources: a marine recreation case." Ocean & Coastal Management **27**(1-2): 129-141.

Kinzig, A. P., S. W. Pacala, et al., Eds. (2002). Functional consequences of biodiversity : empirical progress and theoretical extensions, Princeton University Press.

Klein, R. J. T. and I. J. Bateman (1998). "The recreational value of Cley Marshes nature reserve: An argument against managed retreat?" Journal Of The Chartered Institution Of Water And Environmental Management **12**(4): 280-285.

Kling, C. L. and J. A. Herriges (1995). "An Empirical-Investigation of the Consistency of Nested Logit-Models with Utility Maximization." American Journal of Agricultural Economics **77**(4): 875-884.

Knapp, A. K. and M. D. Smith (2001). Variation Among Biomes in Temporal Dynamics of Aboveground Primary Production. **291:** 481-484.

Kneese, A. (1984). Measuring the Benefits of Clean Air and Water. Washington, DC, Resources for the Future.

Koo, B., P. G. Pardey, et al. (2004). Saving Seeds: The Economics of Conserving Crop Genetic Resources Ex Situ in the Future Harvest Centres of the CGIAR. Wallingford, UK, CABI Publishing.

Kremen, C. and T. Ricketts (2000). "Global perspectives on pollination disruptions." Conservation Biology **14**(5): 1226-1228.

Kremen, C., N. M. Williams, et al. (2002). "Crop pollination from native bees at risk from agricultural intensification." Proceedings of the National Academy of Sciences of the United States of America **99**(26): 16812-16816.

Kriesel, W., A. Keeler, et al. (2004). "Financing beach improvements: Comparing two approaches on the Georgia coast." Coastal Management **32**(4): 433-447.

Kristrom, B. and P. Riera (1996). "Is the income elasticity of environmental improvements less than one?" Environmental and Resource Economics **7**(1): 45-55.

Krutilla, J. V. (1967). "Conservation reconsidered." The American Economic Review **57**(4): 777-789.

Kumar, B. M. and P. K. R. Nair (2004). "The enigma of tropical homegardens."

Agroforestry Systems **61**(1): 135-152.

Lawton, J. H. (1998). "Pigeons, peregrines and people." Oikos **83**(2): 209-211.

Le Goffe, P. (1995). "The Benefits of Improvements in Coastal Water Quality: A Contingent Approach." Journal of Environmental Management **45**: 305-317.

Lee, C. K. and S. Y. Han (2002). "Estimating the use and preservation values of national parks' tourism resources using a contingent valuation method." Tourism Management **23**(5): 531-540.

Leemans, R. and W. Cramer (1991). The IIASA database for mean monthly values of temperature, precipitation and cloudiness on a global terrestrial grid., Laxenburg, Austria, International Institute of Applied Systems Analyses.

Leggett, C. G. and N. E. Bockstael (2000). "Evidence of the effects of water quality on residential land prices." Journal of Environmental Economics and Management **39**(2): 121-144.

Lieth, H. F. H. (1978). Primary patterns of production in the biosphere. . New York, Academic Press.

Limburg, K. E., R. V. O'Neill, et al. (2002). "Complex systems and valuation." Ecological Economics **41**(3): 409-420.

Lindsay, B. E., J. M. Halstead, et al. (1992). "Factors Influencing the Willingness to Pay for Coastal Beach Protection." Coastal Management **20**(3): 291-302.

Loomis, J. (1999). Contingent valuation methodology and the US institutional framework. Valuing environmental preferences. I. Bateman and M. Williams. New York, Oxford University Press: 613-629.

Loomis, J. B. (1988). "The bioeconomic effects of timber harvesting on recreational and commercial salmon and steelhead fishing: a case study of the Siuslaw National Forest." Marine Pollution Bulletin **5**(1): 43-60.

Loomis, J. B. and R. S. Rosenberger (2006). "Reducing barriers in future benefit transfers: Needed improvements in primary study design and reporting." Ecological Economics **60**(2): 343-350.

Loomis, J. B. and D. S. White (1996). "Economic benefits of rare and endangered species: Summary and meta-analysis." Ecological Economics **18**(3): 197-206.

Loreau, M., S. Naeem, et al., Eds. (2002). Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. . Oxford, UK, Oxford University Press.

Lynne, G. D., P. Conroy, et al. (1981). "Economic Valuation of Marsh Areas for Marine Production Processes." Journal of Environmental Economics and Management **8**: 175-186.

MacArthur, R. H. (1955). "Fluctuations of animal populations and a measure of community stability." Ecology **36**: 533-536.

Machado, F. S. and S. Mourato (2002). "Evaluating the multiple benefits of marine water quality improvements: how important are health risk reductions?" Journal Of Environmental Management **65**(3): 239-250.

Mackenzie, J. (1992). "Evaluating Recreation Trip Attributes and Travel Time Via Conjoint-Analysis." Journal of Leisure Research **24**(2): 171-184.

Mader, P., A. Fliessbach, et al. (2002). "Soil fertility and biodiversity in organic farming." Science **296**(5573): 1694-1697.

Martinez-Alier, J., G. Munda, et al. (1998). "Weak comparability of values as a foundation for ecological economics." Ecological Economics **26**(3): 277-286.

Matero, J. and O. Saastamoinen (2007). "In search of marginal environmental valuations - ecosystem services in Finnish forest accounting." Ecological Economics **61**(1): 101-114.

May, R. (1972). "Will a large complex system be stable?" Nature **238**: 413-414.

McCarthy, J. J. a. I. P. o. C. C. W. G. I. (2001). Climate change 2001: impacts, adaptation, and vulnerability: contribution of Working Group II to the third assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK; New York Published for the Intergovernmental Panel on Climate Change [by] Cambridge University Press.

McCauley, D. J. (2006). "Selling out on nature." Nature **443**(7107): 27-28.

McCollum, D. W. (2003). Nonmarket valuation in action. A primer on nonmarket valuation. P. A. Champ, K. J. Boyle and T. C. Brown. Dordrecht, The Netherlands, Kluwer Academic Publishers. **483-535**.

McComb, G., V. Lantz, et al. (2006). "International valuation databases: Overview, methods and operational issues." Ecological Economics **60**(2): 461-472.

McConnell, K. E. and N. Bockstael (2006). Valuing the Environment as a Factor of Production Handbook of Environmental Economics, Vol 2. . K. G. Maler and J. Vincent, Elsevier**:** 621-669.

McNeely, J. A. and S. J. Scherr (2003). *Ecoagriculture: Strategies to Feed the World and Save Biodiversity*. Washington, dc: Island Press. Washington, D.C., Island Press.

Meadows, D. H., D. L. Meadows, et al. (1972). The limits to growth. New York, Potomac Associates.

Meng, E. C. H., M. Smale, et al. (2003). Wheat genetic diversity in China: Measurement and cost. . Agricultural Trade and Policy in China: Issues, Analysis and Implications S. Rozello and D. A. Sumner. Burlington, VT., Ashgate.

Michon, G. and H. de Foresta (1990). Complex agroforestry systems and the conservation of biological diversity. Agroforests in Indonesia: The link between two worlds. . the International Conference on Tropical Biodiversity, Kuala Lumpur, Malaysia., United Selangor Press.

Mikola, J. and H. Setala (1998). "No evidence of trophic cascades in an experimental microbial-based soil food web." Ecology **79**(1): 153-164.

Milestad, R. and S. Hadatsch (2003). "Organic farming and social-ecological resilience: the alpine valleys of Solktaler, Austria." Conservation Ecology **8**(1): 18.

Millennium Ecosystem Assessment (2003). Ecosystems and Human Well-Being: A Framework for Assessment. Washington, DC, Island Press.

Mitchell, R. C. and R. T. Carson (1989). Using surveys to value public goods: the contingent valuation method. Baltimore, John Hopkins University Press.

Moeltner, K., K. J. Boyle, et al. (2007). "Meta-analysis and benefit transfer for resource valuation-addressing classical challenges with Bayesian modeling." Journal of Environmental Economics and Management **53**(2): 250-269.

Montoya, J. M., M. A. Rodriguez, et al. (2003). "Food web complexity and higher-level ecosystem services." Ecology Letters **6**(7): 587-593.

Moran, D. (1999). "Benefits transfer and low flow alleviation: what lessons for environmental valuation in the UK?" Journal of Environmental Planning and Management **42**(3): 425-436.

Morey, E. R., W. D. Shaw, et al. (1991). "A Discrete-Choice Model of Recreational

Participation, Site Choice, and Activity Valuation When Complete Trip Data Are Not Available." Journal of Environmental Economics and Management **20**(2): 181-201.

Mortimer, R., B. Sharp, et al. (1996). "Assessing the conservation value of New Zealand's offshore islands." Conservation Biology **10**(1): 25-29.

Mulder, C. P. H., J. Koricheva, et al. (1999). "Insects affect relationships between plant species richness and ecosystem processes." Ecology Letters **2**(4): 237-246.

Munda, G. (1995). Multicriteria evaluation in a fuzzy environment, Physica-Verlag.

Nabhan, G. P. and S. Buchmann (1997). Services provided by pollinators. . Nature's Services. Societal Dependence on Natural Ecosystems. G. C. Daily. Washington, D.C., Island Press**:** 133-150.

Naeem, S. (1998). "Species redundancy and ecosystem reliability." Conservation Biology **12**(1): 39-45.

Naeem, S. (2001). Experimental validity and ecological scale as tools for evaluating research programs. . Scaling Relationships in Experimental Ecology. R. H. Gardner, W. M. Kemp, V. S. Kennedy and J. E. Petersen. New York, Columbia University Press**:** 223-250.

Naeem, S. (2002). "Ecosystem consequences of biodiversity loss: The evolution of a paradigm." Ecology **83**(6): 1537-1552.

Naeem, S., D. R. Hahn, et al. (2000). "Producer-decomposer co-dependency influences biodiversity effects." Nature **403**(6771): 762-764.

Naeem, S., K. Hakansson, et al. (1996). "Biodiversity and plant productivity in a model assemblage of plant species." Oikos **76**(2): 259-264.

Naeem, S. and S. B. Li (1997). "Biodiversity enhances ecosystem reliability." Nature **390**(6659): 507-509.

Naeem, S., L. J. Thompson, et al. (1994). "Declining Biodiversity Can Alter The Performance Of Ecosystems." Nature **368**(6473): 734-737.

Naeem, S., L. J. Thompson, et al. (1995). "Empirical-Evidence That Declining Species-Diversity May Alter the Performance of Terrestrial Ecosystems." Philosophical Transactions of the Royal Society of London Series B-Biological Sciences **347**(1321): 249-262.

Navrud, S. and G. J. Pruckner (1997). "Environmental valuation -- to use or not to use? a comparative study of the United States and Europe." Environmental & Resource Economics **10**(1): 1-26.

Newell, R. G. and W. A. Pizer (2003). "Discounting the distant future: how much do uncertain rates increase valuations?" Journal of Environmental Economics and Management **46**(1): 52-71.

Niklitschek, M. and J. Leon (1996). "Combining intended demand and yes/no responses in the estimation of contingent valuation models." Journal Of Environmental Economics And Management **31**(3): 387-402.

Nordhaus, W. (2007). "ECONOMICS: Critical Assumptions in the Stern Review on Climate Change." Science **317**(5835): 201-202.

Norgaard, R. B. and C. Bode (1998). "Next, the value of God, and other reactions." Ecological Economics **25**(1): 37-39.

Nunes, P. (2002). "Using factor analysis to identify consumer preferences for the protection of a natural area in Portugal." European Journal Of Operational

Research **140**(2): 499-516.

Nunes, P., L. Rossetto, et al. (2004). "Measuring the economic value of alternative clam fishing management practices in the Venice Lagoon: Results from a conjoint valuation application." Journal Of Marine Systems **51**(1-4): 309-320.

Nunes, P. and J. Van den Bergh (2004). "Can people value protection against invasive marine species? Evidence from a joint TC-CV survey in the Netherlands." Environmental & Resource Economics **28**(4): 517-532.

Odum, E. P. (1953). Fundamentals of ecology. Philadelphia, Saunders.

Odum, E. P. (1979). "Rebuttal of ‘‘economic value of natural coastal wetlands: a critique.’’." Coastal Zone Manage **J 5**: 231-237.

Odum, H. T. (1967). Energetics of world food production. In: Problems of world food supply. President’s Science Advisory Committee Report. Volume 3. Washington, DC, The White House**:** 55–94.

Odum, H. T. (1971). Environment, power, and society. New York, Wiley-Interscience.

Odum, H. T. (1979). "Principle of environmental energy matching for estimating potential value: a rebuttal." Coastal Zone Manage **J 5**: 239-241.

OECD (2002). Handbook of biodiversity valuation-a guide for policy makers, OECD.

Office of Policy Analysis and Environmental Protection Agency (1987). EPA's use of cost benefit analysis: 1981-1986.

Olschewski, R., T. Tscharntke, et al. (2006). "Economic evaluation of pollination services comparing coffee landscapes in Ecuador and Indonesia." Ecology and Society **11**(1).

Orians, G. H., G. M. Brown, et al., Eds. (1990). Preservation and Valuation of Biological

Resources. Seattle, WA., University of Washington Press.

Ostman, O., B. Ekbom, et al. (2001). "Landscape complexity and farming practice influence the condition of polyphagous carabid beetles." Ecological Applications **11**(2): 480-488.

Pacala, S. W. and D. Tilman (2002). The transition from sampling to complementarity. Functional consequences of biodiversity : empirical progress and theoretical extensions. A. P. Kinzig, S. W. Pacala and D. Tilman. Princeton, NJ, Princeton University Press**:** 151-166.

Pain, D. J. and M. W. Pienkowski (1997). Farming and Birds in Europe: The Common Agricultural Policy and Its Implications for Bird Conservation. . Cambridge, UK, Academic Press.

Paine, R. T. (1966). "Food web complexity and species diversity." American Naturalist **100**: 65-75.

Paine, R. T. (2002). Trophic Control of Production in a Rocky Intertidal Community. **296:** 736-739.

Park, T., J. M. Bowker, et al. (2002). "Valuing snorkeling visits to the Florida Keys with stated and revealed preference models." Journal of Environmental Management **65**(3): 301-312.

Parsons, E. C. M., C. A. Warburton, et al. (2003). "The value of conserving whales: the impacts of cetacean-related tourism on the economy of rural West Scotland." Aquatic Conservation-Marine And Freshwater Ecosystems **13**(5): 397-415.

Parsons, G. R. and Y. R. Wu (1991). "The Opportunity Cost Of Coastal Land-Use Controls - An Empirical-Analysis." Land Economics **67**(3): 308-316.

Pearce, D. (1998). "Auditing the Earth." Environment **40**: 23-28.

Pearce, D. W. and T. Seccombe-Hett (2000). "Economic valuation and environmental decision-making in Europe." Environmental Science & Technology **34**(8): 1419-1425.

Perfecto, I., J. Vandermeer, et al. (1997). "Arthropod biodiversity loss and the transformation of a tropical agro-ecosystem." Biodiversity and Conservation **6** 935-945.

Perrings, C. (1995). Biodiversity conservation as insurance. The economics and ecology of biodiversity. T. Swanson. New York, Cambridge University Press**:** 91-98.

Perrings, C. (2006). "Ecological Economics after the Millennium Assessment." International Journal of Ecological Economics & Statistics **6**: 8-23.

Petchey, O. L., P. J. Morin, et al. (2002). Contributions of aquatic model systems to our understanding of biodiversity and ecosystem functioning. Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. . M. Loreau, S. Naeem and P. Inchausi. Oxford, UK, Oxford University Press**:** 127-138.

Pfisterer, A. B. and B. Schmid (2002). "Diversity-dependent production can decrease the stability of ecosystem functioning." Nature **416**(6876): 84-86.

Pimm, S. L. (1997). "The value of everything." Nature **387**(6630): 231-232.

Pinto-Correia, T. (2000). "Future development in Portuguese rural areas: how to manage agricultural support for landscape conservation?" Landscape and Urban Planning **50**(1-3): 95-106.

Prato, T. (1999). "Multiple attribute decision analysis for ecosystem management." Ecological Economics **30**(2): 207-222.

Pretty, J. and H. Ward (2001). "Social capital and the environment." World Development **29**(2): 209-227.

Priess, J. A., M. Mimler, et al. (2007). "Linking deforestation scenarios to pollination services and economic returns in coffee agroforestry systems." Ecological Applications **17**(2): 407-417.

Ricketts, T., E. Dinerstein, et al. (1999). Terrestrial ecoregions of north America, Island Press.

Ricketts, T. H., G. C. Daily, et al. (2001). "Countryside biogeography of moths in a fragmented landscape: Biodiversity in native and agricultural habitats." Conservation Biology **15**(2): 378-388.

Ricketts, T. H., G. C. Daily, et al. (2004). "Economic value of tropical forest to coffee production." Proceedings of the National Academy of Sciences of the United States of America **101**(34): 12579-12582.

Ridker, R. G. and J. A. Henning (1967). "the determinants of residential property values with special reference to air pollution." The Review of Economics and Statistics **49**(2): 246-257.

Robinson, R. A., J. D. Wilson, et al. (2001). "The importance of arable habitat for farmland birds in grassland landscapes (vol 38, pg 1059, 2001)." Journal of Applied Ecology **38**(6): 1386-1386.

Rojstaczer, S., S. M. Sterling, et al. (2001). Human Appropriation of Photosynthesis Products. **294:** 2549-2552.

Rollins, K. and A. Lyke (1998). "The Case for Diminishing Marginal Existence Values." Journal of Environmental Economics and Management **36**(3): 324-344.

Ropke, I. (2004). "The early history of modern ecological economics." Ecological Economics **50**: 293-314.

Rosenberger, R. S. and J. B. Loomis (2000). Benefit transfer of ourdoor recreation use values, USDA Forest Service.

Rosenberger, R. S. and J. B. Loomis (2000). "Using meta-analysis for benefit transfer: In-sample convergent validity tests of an outdoor recreation database." Water Resources Research **36**(4): 1097-1107.

Rosenberger, R. S. and T. D. Stanley (2006). "Measurement, generalization, and publication: Sources of error in benefit transfers and their management." Ecological Economics **60**(2): 372-378.

Rowe, R. D., W. D. Shaw, et al. (1992). Nestucca Oil Spill. Natural Resource Damages. K. Ward and J. Duffield. New York, Wiley and Sons.

Running, S. W. and J. C. Coughlan (1988). "A General-Model of Forest Ecosystem Processes for Regional Applications .1. Hydrologic Balance, Canopy Gas-Exchange and Primary Production Processes." Ecological Modelling **42**(2): 125-154.

Schimid, B., J. Joshi, et al. (2002). Empirical evidence for biodiversity–ecosystem functioning relationships. Functional consequences of biodiversity : empirical progress and theoretical extensions. A. P. Kinzig, S. W. Pacala and D. Tilman, Princeton University Press**:** 120-150.

Schlapfer, F. (2006). "Survey protocol and income effects in the contingent valuation of public goods: A meta-analysis." Ecological Economics **57**(3): 415-429.

Schmalensee, R. (1976). "Another look at social valuation of input price changes."

American Economic Review **66**(1): 239-243.

Schulze, E. D. and H. Mooney (1993). Biodiversity and ecosystem function. Berlin, Springer-Verlag.

Scoones, I. (1999). "New ecology and the social sciences: What prospects for a fruitful engagement?" Annual Review of Anthropology **28**: 479-507.

Shabman, L. and M. K. Bertelson (1979). "The use of development value estimates for coastal wetland permit decisions." Land Economics **55**(2): 213-222.

Shabman, L. A. and S. S. Batie (1978). "Economic value of natural coastal wetlands: a critique." Coastal Zone Manage **J 4**(3): 231-247.

Shrestha, R. K. and J. B. Loomis (2003). "Meta-analytic benefit transfer of outdoor recreation economic values: Testing out-of-sample convergent validity." Environmental & Resource Economics **25**(1): 79-100.

Silberman, J., D. A. Gerlowski, et al. (1992). "Estimating Existence Value for Users and Nonusers of New-Jersey Beaches." Land Economics **68**(2): 225-236.

Silberman, J. and M. Klock (1988). "The Recreation Benefits of Beach Renourishment." Ocean & Shoreline Management **11**: 73-90.

Silva, P. and S. Pagiola (2003). A review of the valuation of environmental costs and benefits in world bank projects. Washington, D.C.

Smale, M., Ed. (2005). Valuing Crop Biodiversity: On-Farm Genetic Resources and Economic Change. Wallingford, UK CABI Publishing.

Smale, M., J. Hartell, et al. (1998). "The contribution of genetic resources and diversity to wheat production in the Punjab of Pakistan." American Journal of Agricultural Economics **80**(3): 482-493.

Smith, V. K. (1984). Environmental policy making under executive order 12291: an introduction. Environmental policy under Reagan's executive Order. V. K. Smith, The University of North Carolina Press**:** 3-40.

Smith, V. K. (1992). "On Separating Defensible Benefit Transfers from Smoke and Mirrors." Water Resources Research **28**(3): 685-694.

Smith, V. K. (1993). "Nonmarket Valuation of Environmental Resources - an Interpretive Appraisal." Land Economics **69**(1): 1-26.

Smith, V. K. (2000). "JEEM and non-market valuation: 1974-1998." Journal of Environmental Economics and Management **39**(3): 351-374.

Smith, V. K. and Y. Kaoru (1990). "Signals or Noise - Explaining the Variation in Recreation Benefit Estimates." American Journal of Agricultural Economics **72**(2): 419-433.

Smith, V. K. and Y. Kaoru (1990). "What Have We Learned since Hotelling Letter - a Metaanalysis." Economics Letters **32**(3): 267-272.

Smith, V. K. and L. L. Osborne (1996). "Do contingent valuation estimates pass a "scope" test? A meta-analysis." Journal of Environmental Economics and Management **31**(3): 287-301.

Smith, V. K. and S. K. Pattanayak (2002). "Is meta-analysis a Noah's ark for non-market valuation?" Environmental & Resource Economics **22**(1-2): 271-296.

Smith, V. K., X. Zhang, et al. (1997). "Marine debris, beach quality, and non-market values." Environmental and Resource Economics **10**: 223-247.

Soderstrom, B., S. Kiema, et al. (2003). "Intensified agricultural land-use and bird conservation in Burkina Faso." Agriculture Ecosystems & Environment **99**(1-3):

113-124.

Solomon, B. D., C. M. Corey-Luse, et al. (2004). "The Florida manatee and eco-tourism: toward a safe minimum standard." Ecological Economics **50**(1-2): 101-115.

Southwick, E. E. and L. Southwick (1992). "Estimating the Economic Value of Honey-Bees (Hymenoptera, Apidae) as Agricultural Pollinators in the United-States." Journal of Economic Entomology **85**(3): 621-633.

Stanley, T. D. (2001). "Wheat from chaff: Meta-analysis as quantitative literature review." Journal of Economic Perspectives **15**(3): 131-150.

Stanley, T. D. (2005). "Beyond publication bias." Journal of Economic Surveys **19**(3): 309-345.

Steffan-Dewenter, I. (2003). "Importance of habitat area and landscape context for species richness of bees and wasps in fragmented orchard meadows." Conservation Biology **17**(4): 1036-1044.

Steffan-Dewenter, I., U. Munzenberg, et al. (2002). "Scale-dependent effects of landscape context on three pollinator guilds." Ecology **83**(5): 1421-1432.

Stein, T. V., D. H. Anderson, et al. (1999). "Using stakeholders' values to apply ecosystem management in an upper midwest landscape." Environmental Management **24**(3): 399-413.

Stern, N. and C. Taylor (2007). "ECONOMICS: Climate Change: Risk, Ethics, and the Stern Review." Science **317**(5835): 203-204.

Stevens, T. H., R. Belkner, et al. (2000). "Comparison of contingent valuation and conjoint analysis in ecosystem management." Ecological Economics **32**(1): 63-74.

Sumaila, U. R. and C. Walters (2005). "Intergenerational discounting: a new intuitive approach." Ecological Economics **52**(2): 135-142.

Symstad, A. J., F. S. Chapin, et al. (2003). "Long-term and large-scale perspectives on the relationship between biodiversity and ecosystem functioning." Bioscience **53**(1): 89-98.

Tews, J., U. Brose, et al. (2004). "Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures." Journal of Biogeography **31**(1): 79-92.

Thies, C. and T. Tscharntke (1999). "Landscape structure and biological control in agroecosystems." Science **285**(5429): 893-895.

Thrupp, L. A. (1997). Linking Biodiversity and Agriculture: Challenges and Opportunities for Sustainable Food Security. Washington, DC, World Resource Institute.

Tilman, D. and J. A. Downing (1994). "Biodiversity and Stability in Grasslands." Nature **367**(6461): 363-365.

Tilman, D., J. Knops, et al. (2002). Experimental and observational studies of diversity, productivity, and stability. Functional consequences of biodiversity : empirical progress and theoretical extensions. A. P. Kinzig, S. W. Pacala and D. Tilman, Princeton University Press: 42-70.

Tilman, D., J. Knops, et al. (1997). "The influence of functional diversity and composition on ecosystem processes." Science **277**(5330): 1300-1302.

Tilman, D. and C. Lehman (2002). Biodiversity, composition and ecosystem processes: theory and concepts. Functional consequences of biodiversity : empirical progress and theoretical extensions. A. P. Kinzig, S. W. Pacala and D. Tilman, Princeton

University Press: 9-41.

Tilman, D., C. L. Lehman, et al. (1997). "Plant diversity and ecosystem productivity: Theoretical considerations." Proceedings of the National Academy of Sciences of the United States of America **94**(5): 1857-1861.

Tilman, D., D. Wedin, et al. (1996). "Productivity and sustainability influenced by biodiversity in grassland ecosystems." Nature **379**(6567): 718-720.

Tol, R. S. J. (1999). "The marginal costs of greenhouse gas emissions." Energy Journal **20**(1): 61-81.

Tonhasca, A. and B. R. Stinner (1991). "Effects of Strip Intercropping and No-Tillage on Some Pests and Beneficial Invertebrates of Corn in Ohio." Environmental Entomology **20**(5): 1251-1258.

Townend, I. (In press). "Marine Science for strategic planning and management: the requirement for estuaries." Marine Policy.

Troy, A. and M. A. Wilson (2006). "Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer." Ecological Economics **60**(2): 435-449.

Tsuge, T. and T. Washida (2003). "Economic valuation of the Seto Inland Sea by using an Internet CV survey." Marine Pollution Bulletin **47**(1-6): 230-236.

Turner, D. P., W. D. Ritts, et al. (2003). "Scaling Gross Primary Production (GPP) over boreal and deciduous forest landscapes in support of MODIS GPP product validation." Remote Sensing of Environment **88**(3): 256-270.

Turner, R. K. (2000). "Integrating natural and socio-economic science in coastal management." Journal of Marine Systems **25**(3-4): 447-460.

Turner, R. K., J. Paavola, et al. (2003). "Valuing nature: lessons learned and future research directions." Ecological Economics **46**(3): 493-510.

Turner, R. K., S. Subak, et al. (1996). "Pressures, Trends and Impacts in Coastal Zones: Interactions Between Socioeconomic and Natural Systems." Environmental Management **20**(2): 159-173.

Udziela, M. K. and L. L. Bennett (1997). "Contingent valuation of an urban salt marsh restoration." Yale Forestry and Environmental Studies Bulletin **100**: 41-61.

US National Research Council (2005). Valuing Ecosystem Services: Toward Better environmental Decision-Making. Washington, D.C., The National Academies Press.

van den Bergh, J. C. J. M., K. Button, et al. (1997). Meta-analysis in environmental economics, Kluwer Academic Publishers.

Vatn, A. and D. W. Bromley (1994). "Choices without Prices without Apologies." Journal of Environmental Economics and Management **26**(2): 129-148.

Vitousek, P. M., P. R. Ehrlich, et al. (1986). "Human Appropriation of the Products of Photosynthesis." Bioscience **36**(6): 368-373.

Vitousek, P. M., H. A. Mooney, et al. (1997). "Human domination of Earth's ecosystems." Science **277**(5325): 494-499.

Walker, B., S. Carpenter, et al. (2002). "Resilience management in social-ecological systems: a working hypothesis for a participatory approach." Conservation Ecology **6**(1).

Walker, B. H. (1992). "Biodiversity and Ecological Redundancy." Conservation Biology **6**(1): 18-23.

Walker, B. H. and W. E. Steffen (1996). Global change and terrestrial ecosystems. Cambridge, Cambridge University Press.

Walker, D. J. and D. L. Young (1986). "The Effect of Technical Progress on Erosion Damage and Economic Incentives for Soil Conservation." Land Economics **62**(1): 83-93.

Walsh, R. G., D. M. Johnson, et al. (1989). "Issues in nonmarket valuation and policy application: a retrospective glance." Western Journal of Agricultural Economics **14**: 178-188.

Walsh, R. G., D. M. Johnson, et al. (1992). "Benefit Transfer of Outdoor Recreation Demand Studies, 1968- 1988." Water Resources Research **28**(3): 707-713.

Wardle, D. A. and O. Zackrisson (2005). "Effects of species and functional group loss on island ecosystem properties." Nature **435**(7043): 806-810.

Wardle, D. A., O. Zackrisson, et al. (1997). "Biodiversity and ecosystem properties - Response." Science **278**(5345): 1867-1869.

Wardle, D. A., O. Zackrisson, et al. (1997). "The influence of island area on ecosystem properties." Science **277**(5330): 1296-1299.

Weisbrod, B. A. (1964). "Collective consumption services of individual consumption goods." Quarterly Journal of Economics **77**(Aug.): 71-77.

Weitzman, M. L. (1998). "Why the far-distant future should be discounted at its lowest possible rate." Journal of Environmental Economics and Management **36**(3): 201-208.

Whitehead, J. C. (1993). "Total economic values for coastal and marine wildlife: specification, validity, and valuation issues." Marine Resource Economics

8(119-132).

Whitehead, J. C., G. C. Blomquist, et al. (1995). "Assessing the Validity and Reliability of Contingent Values - a Comparison of on-Site Users, Off-Site Users, and Nonusers." Journal of Environmental Economics and Management **29**(2): 238-251.

Whitehead, J. C., G. C. Blomquist, et al. (1998). "Construct validity of dichotomous and polychotomous choice contingent valuation questions." Environmental & Resource Economics **11**(1): 107-116.

Whitehead, J. C., T. C. Haab, et al. (2000). "Measuring recreation benefits of quality improvements with revealed and stated behavior data." Resource and Energy Economics **22**(4): 339-354.

Widawsky, D. and S. Rozelle (1998). Varietal diversity and yield variability in Chinese rice production. . Farmers, Gene Banks, and Crop Breeding. M. Smale. Boston, Kluwer.

Willms, W. D., J. F. Dormaar, et al. (2002). "Response of the mixed prairie to protection from grazing." Journal of Range Management **55**(3): 210-216.

Wilson, E. O. (1988). Biodiversity. Washington, D.C., National Academy Press.

Wilson, M. and S. Liu (2007). Evaluating the Non-Market Value of Ecosystem Goods and Services provided by Coastal and Nearshore Marine System. . Ecological Economics of the Oceans and Coasts. . G. Patterson and Glavovic.

Wilson, M. and A. Troy (2003). Accounting for the economic value of ecosystem services in Massachusetts. Loosing Ground: at what cost. K. Breunig. Boston, Massachusetts Audubon Society.

Wilson, M., A. Troy, et al. (2004). The Economic Geography of Ecosystem Goods and Services:Revealing the monetary value of landscapes through transfer methods and Geographic Information Systems. Cultural Landscapes and Land Use. M. Dietrich and V. D. Straaten, Kluwer Academic.

Wilson, M. A. and S. R. Carpenter (1999). "Economic Valuation of Freshwater Ecosystem Services in the United States 1971-1997." Ecological Applications **9**(3): 772-783.

Wilson, M. A., R. Costanza, et al. (2005). Intergrated Assessment and Valuation of Ecosystem Goods and Services Provided by Coastal Systems. The Intertidal Ecosystem. J. G. Wilson. Dublin, Ireland, Royal Irish Academy Press**:** 1-24.

Wilson, M. A. and J. P. Hoehn (2006). "Valuing environmental goods and services using benefit transfer: The state-of-the art and science." Ecological Economics **60**(2): 335-342.

Wilson, M. A. and R. B. Howarth (2002). "Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation." Ecological Economics **41**(3): 431-443.

Wood, S. K. and S. J. Scherr (2000). Pilot Analysis of Global Ecosystems: Agroecosystems. Washington, DC: International Food Policy Research Institute and World Resources Institute. Washington, DC, International Food Policy Research Institute and World Resources Institute.

Woodward, R. T. and Y. S. Wui (2001). "The economic value of wetland services: a meta-analysis." Ecological Economics **37**(257-270).

Wright, J. P., C. G. Jones, et al. (2002). "An ecosystem engineer, the beaver, increases species richness at the landscape scale." Oecologia **132**(1): 96-101.

Yachi, S. and M. Loreau (1999). "Biodiversity and ecosystem productivity in a fluctuating environment: The insurance hypothesis." Proceedings of the National Academy of Sciences of the United States of America **96**(4): 1463-1468.

Yeo, B. H. (2002). Valuing a Marine Park in Malaysia. Valuing the Environment in Developing Countries: Case Studies,. D. Pearce, C. Pearce and C. Palmer. Cheltenham, UK; Northampton, MA, USA, Edward Elgar**:** 311-324.

Zhu, Y. Y., H. R. Chen, et al. (2000). "Genetic diversity and disease control in rice." Nature **406**(6797): 718-722.

Appendix A. New Jersey Value-Transfer Detailed Report

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Beach								
	Disturbance prevention							
		Pompe, J. J. and Rinehart, J. R.-1995	HP			$33,738	$33,738	$33,738
		Parsons, G. R. and Powell, M.-2001-2001	HP			$20,814	$20,814	$20,814
						Disturbance prevention	**$27,276**	**$27,276**
	Aesthetic & Recreational							
		Taylor, L. O. and Smith, V. K.-2000	HP	$392	$1,058		$725	$725
		Silberman, J., Gerlowski, D. A. and Williams, N. A.1992	CV			$20,680	$20,680	$20,680
		Kline, J. D. and Swallow, S. K.-1998	CV	$33,051	$42,654		$37,853	$37,853
		Edwards, S. F. and Gable, F. J.-1991	HP			$131	$131	$131
						Aesthetic & Recreational	**$14,847**	**$10,703**
	Cultural & Spiritual							
		Taylor, L. O. and Smith, V. K.-2000	HP			$24	$24	$24
						Cultural & Spiritual	**$24**	**$24**
						Beach Total	**$42,147**	**$38,002**

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
						2004 dollars per acre/year		
Cropland								
	Pollination							
		Southwick, E. E. and Southwick, L.-1992	DM	$2	$8		$5	$5
		Robinson, W. S., Nowogrodzki, R. and Morse, R. A.-1989	AC			$11	$11	$11
						Pollination	**$8**	**$8**
	Aesthetic & Recreational							
		Bergstrom, J., Dillman, B. L. and Stoll, J. R.-1985	CV			$26	$26	$26
		Alvarez-Farizo, B., Hanley, N., Wright, R. E. and MacMillan, D.	CV			$4	$4	$4
						Aesthetic & Recreational	**$15**	**$15**
						Cropland Total	**$23**	**$23**
Estuary								
	Water supply							
		Whitehead, J. C., Hoban, T. L. and Clifford, W. B.-1997	CV	$6	$21		$13	$13
		Leggett, C. G. and Bockstael, N. E.-2000	HP			$40	$40	$40
		Bocksteal, N. E., McConnell, K. E. and Strand, I. E.1989	CV	$67	$120		$94	$94
						Water supply	**$49**	**$40**

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
						2004 dollars per acre/year		
	Refugium function							
		Johnston, R. J. et. al.-2002	MP			$412	$412	$412
		Johnston, R. J. et. al.-2002	MP			$1,298	$1,298	$1,298
Estuary, cont.		Johnston, R. J. et. al.-2002	MP			$82	$82	$82
	.	Farber, S. and Costanza, R.-1987	MP			$15	$15	$15
		Farber, S. and Costanza, R.-1987	MP			$11	$11	$11
						Refugium function	**$364**	**$82**
	Aesthetic & Recreational							
		Whitehead, J. C., Hoban, T. L. and Clifford, W. B.-1997	CV	$1	$5		$3	$3
		Whitehead, J. C., Hoban, T. L. and Clifford, W. B.-1997	CV	$9	$81		$45	$45
		Morey, E. R., Shaw, W. D. and Rowe, R. D.	TC			$68	$68	$68
		Johnston, R. J. et. al.-2002	TC			$148	$148	$148
		Johnston, R. J. et. al.-2002	TC			$289	$289	$289
		Johnston, R. J. et. al.-2002	TC			$333	$333	$333
		Johnston, R. J. et. al.-2002	TC			$158	$158	$158
		Johnston, R. J. et. al.-2002	TC			$219	$219	$219
		Johnston, R. J., Opaluch, J. J.,	CV			$1,462	$1,462	$1,462

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
		Grigalunas, T. A. and Mazzotta, M. J.						
				Aesthetic & Recreational			**$303**	**$158**
				Estuary Total			**$715**	**$281**
Forest								
	Gas & Climate regulation							
		Pimentel, D.-1998	AC			$13	$13	$13
		Tol, R. S. J.	MP			$57	$57	$57
		Tol, R. S. J.	MP			$302	$302	$302
		Schauer, M. J.	MP			$318	$318	$318
		Schauer, M. J.	MP			$23	$23	$23
		Roughgarden, T. and Schneider, S. H.	MP		$184	$39	$39	$39
		Reilly, J. M. and Richards, K. R.	MP			$49	$49	$49
		Reilly, J. M. and Richards, K. R.	MP			$42	$42	$42
		Reilly, J. M. and Richards, K. R.	MP			$20	$20	$20
		Reilly, J. M. and Richards, K. R.	MP			$14	$14	$14
		Plambeck, E. L. and Hope, C.	MP	$371	$933	$419	$419	$419
		Plambeck, E. L. and Hope, C.	MP	$10	$46	$20	$20	$20
		Nordhaus, W. D. and Popp, D.	MP	$0.04	$32	$11	$11	$11

Land Cover	Author(s)	Method	Min	Max	Single Value	Mean	Median
			2004 dollars per acre/year				
	Nordhaus, W. D. and Popp, D.	MP	$1	$42	$6	$6	$6
	Nordhaus, W. D. and Yang, Z. L.	MP			$0.23	$0.23	$0.23
	Nordhaus, W. D. and Yang, Z. L.	MP			$6	$6	$6
	Nordhaus, W. D.	MP			$5	$5	$5
	Nordhaus, W. D.	MP	$2	$15	$7	$7	$7
	Nordhaus, W. D.	MP	$0.31	$2	$1	$1	$1
	Nordhaus, W. D.	MP	$8	$66	$31	$31	$31
Forest, cont.	Newell, R. G. and Pizer, W. A.	MP	$7	$23		$15	$15
	Newell, R. G. and Pizer, W. A.	MP	$10	$34		$22	$22
	Maddison, D.	MP			$16	$16	$16
	Hope, C. and Maul, P.	MP	$11	$43	$28	$28	$28
	Fankhauser, S.	MP	$23	$66	$40	$40	$40
	Fankhauser, S.	MP	$5	$37	$17	$17	$17
	Fankhauser, S.	MP	$6	$43	$19	$19	$19
	Azar, C. and Sterner, T.	MP			$66	$66	$66
	Azar, C. and Sterner, T.	MP			$10	$10	$10
	Azar, C. and Sterner, T.	MP			$202	$202	$202
	Azar, C. and Sterner, T.	MP			$30	$30	$30
			Gas & Climate regulation			**$60**	**$20**

Land Cover	Author(s)	Method	Min	Max	Single Value	Mean	Median
			2004 dollars per acre/year				
Water supply							
	Loomis, J. B.	TC	$9	$9		$9	$9
					Water supply	**$9**	**$9**
Pollination							
	Hougner, C.	RC	$59	$265		$162	$162
					Pollination	**$162**	**$162**
Refugium function and Wildlife conservation							
	Shafer, E. L. et. al.-1993	CV			$3	$3	$3
	Kenyon, W. and Nevin, C.-2001	CV			$426	$426	$426
	Haener, M. K. and Adamowicz, W. L.-2000	CV	$1	$7		$4	$4
	Amigues, J. P., et. al.-2002	CV	$55	$208		$132	$132
	Amigues, J. P., et. al.-2002	CV	$1,140	$2,158		$1,649	$1,649
	Garrod, G. D. and Willis, K. G.	CV			$15	$15	$15
	Garrod, G. D. and Willis, K. G.	CV	$3,101	$3,383		$3,242	$3,242
	Garrod, G. D. and Willis, K. G.	CV	$1,817	$2,003		$1,910	$1,910
					Refugium function	**$923**	**$279**
Aesthetic & Recreational							
	Willis, K. G.-1991	TC	$89	$162		$126	$126
	Willis, K. G.-1991	TC	$20	$35		$28	$28

Land Cover	Author(s)	Method	Min	Max	2004 dollars per acre/year Single Value	Mean	Median
	Willis, K. G.-1991	TC	$8	$15		$12	$12
	Willis, K. G.-1991	TC	$5	$5		$5	$5
	Willis, K. G.-1991	TC	$0	$1		$1	$1
	Willis, K. G. and Garrod, G. D.-1991	TC			$4	$4	$4
	Shafer, E. L., et. al.-1993	CV			$459	$459	$459
	Prince, R. and Ahmed, E.-1989	CV	$1	$2		$1	$1
	Maxwell, S.-1994	CV			$10	$10	$10
.	Haener, M. K. and Adamowicz, W. L.2000	CV			$0	$0	$0
	Boxall, P. C., McFarlane, B. L. and Gartrell, M.-1996	TC			$0	$0	$0
	Bishop, K.-1992	CV			$543	$543	$543
	Bishop, K.-1992	CV			$485	$485	$485
Forest, cont.	Bennett, R., et. al.-1995	CV			$144	$144	$144
					Aesthetic & Recreational	**$130**	**$11**
					Forest Total	**$1,283**	**$481**
Freshwater Wetland							
Water regulation							
	Thibodeau, F. R. and Ostro, B. D.-1981	AV			$5,957	$5,957	$5,957
					Water regulation	**$5,957**	**$5,957**

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
	Water supply							
		Pate, J. and Loomis, J.-1997	CV			$3,066	$3,066	$3,066
		Lant, C. L. and Roberts, R. S.-1990	CV	$0	$0		$0	$0
		Lant, C. L. and Tobin, G.-1989	CV			$170	$170	$170
		Lant, C. L. and Tobin, G.-1989	CV			$1,868	$1,868	$1,868
		Hayes, K. M., Tyrrell, T. J. and Anderson, G.-1992	CV	$1,097	$1,706		$1,401	$1,401
		Creel, M. and Loomis, J.-1992	TC			$462	$462	$462
	.					***Water supply***	**$1,161**	**$932**
	Refugium function and Wildlife conservation							
		Vankooten, G. C. and Schmitz, A.-1992	CV			$5	$5	$5
						Refugium function	**$5**	**$5**
Freshwater wetland, cont.	*Aesthetic & Recreational*							
		Whitehead, J. C.-1990	CV	$890	$1,790		$1,340	$1,340
		Thibodeau, F. R. and Ostro, B. D.-1981	CV			$559	$559	$559
		Thibodeau, F. R. and Ostro, B. D.-1981	TC	$27	$86		$56	$56

Land Cover		Author(s)	Method	2004 dollars per acre/year Min	Max	Single Value	Mean	Median
		Mahan, B. L., Polasky, S. and Adams, R. M.-2000	TC			$30	$30	$30
		Hayes, K. M., Tyrrell, T. J. and Anderson, G.-1992	CV	$1,033	$1,975		$1,504	$1,504
		Doss, C. R. and Taff, S. J.-1996	TC			$3,942	$3,942	$3,942
		Doss, C. R. and Taff, S. J.-1996	TC			$3,568	$3,568	$3,568
						Aesthetic & Recreational	**$1,571**	**$1,340**
				Freshwater Wetland Total			**$8,695**	**$8,234**
Open Fresh Water								
	Water supply							
		Ribaudo, M. and Epp, D. J.-1984	TC	$567	$719		$643	$643
		Piper, S.-1997	CV			$28	$28	$28
		Henry, R., Ley, R. and Welle, P.1998	CV			$366	$366	$366
		Croke, K., Fabian, R. and Brenniman, G.-1986	CV			$482	$482	$482
		Bouwes, N. W. and Scheider, R.-1979	TC			$526	$526	$526
						Water supply	**$409**	**$482**
Open freshwater, cont.	*Aesthetic & Recreational*							
		Young, C. E. and Shortle, J. S.	HP			$70	$70	$70

Land Cover	Author(s)	Method	Min	Max	2004 dollars per acre/year Single Value	Mean	Median
	Ward, F. A., Roach, B. A. and Henderson, J. E.-1996	TC	$17	$1,635		$826	$826
	Shafer, E. L. et. al. -1993	CV			$83	$83	$83
	Shafer, E. L. et. al. -1993	TC			$470	$470	$470
	Shafer, E. L. et. al. -1993	TC			$938	$938	$938
	Piper, S.-1997	TC			$205	$205	$205
	Patrick, R.,et. al. -1991	TC	$1	$22		$12	$12
	Kreutzwiser, R.-1981	TC			$154	$154	$154
	Kealy, M. J. and Bishop, R. C.-1986	TC			$11	$11	$11
	Cordell, H. K. and Bergstrom, J. C.-1993	CV	$162	$679		$420	$420
	Cordell, H. K. and Bergstrom, J. C.-1993	CV	$115	$242		$179	$179
	Cordell, H. K. and Bergstrom, J. C.-1993	CV	$241	$682		$462	$462
	Cordell, H. K. and Bergstrom, J. C.-1993	CV	$326	$1,210		$768	$768
	Burt, O. R. and Brewer, D.-1971	TC			$393	$393	$393
					Aesthetic & Recreational	**$356**	**$299**
					Open Fresh Water Total	**$765**	**$781**

Pasture

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
						2004 dollars per acre/year		
	Gas & Climate regulation							
		Sala, O. E. and Paruelo, F. M.	MP	$4	$10	$5	$5	$5
				Gas & Climate regulation			**$5**	**$5**
	Soil formation							
		Pimentel, D.-1998	DM			$6	$6	$6
						Soil formation	**$6**	**$6**
	Aesthetic & Recreational							
		Boxall, P. C.-1995	TC			$0.03	$0.03	$0.03
		Alvarez-Farizo, B., Hanley, N., Wright, R. E. and MacMillan, D.				$1	$1	$1
				Aesthetic & Recreational			**$1**	**$1**
						Pasture Total	**$12**	**$12**
Riparian Buffer								
	Disturbance prevention							
		Rein, F. A.-1999	TC	$45	$201		$123	$123
		Rein, F. A.-1999	TC	$6	$99		$53	$53
				Disturbance prevention			**$88**	**$88**
	Water supply							
		Rich, P. R. and Moffitt, L. J.-1982	HP			$4	$4	$4

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Riparian buffer, cont.		Rein, F. A.-1999	AC	$36	$158		$97	$97
		Oster, S.-1977	CV			$13	$13	$13
		Mathews, L. G., Homans, F. R. and Easter, K. W.-2002	CRS			$11,089	$11,089	$11,089
		Kahn, J. R. and Buerger, R. B.	TC	$0.15	$0.77		$0.46	$0.46
		Kahn, J. R. and Buerger, R. B.	TC	$3	$3		$6	$6
		Gramlich, F. W.-1977	CV			$188	$188	$188
		Danielson, L., et. al.-1995	CV			$4,095	$4,095	$4,095
		Berrens, R. P., Ganderton, P. and Silva, C. L.-1996	CV			$1,794	$1,794	$1,794
						Water supply	**$1,921**	**$97**
	Aesthetic & Recreational							
		Sanders, L. D., Walsh, R. G. and Loomis, J. B.-1990	CV			$1,957	$1,957	$1,957
		Rein, F. A.-1999	DM	$26	$113		$69	$69
		Mullen, J. K. and Menz, F. C.-1985	TC			$328	$328	$328
		Kulshreshtha, S. N. and Gillies, J. A.-1993	HP			$43	$43	$43
		Greenley, D., Walsh, R. G. and Young, R. A.-1981	CV			$7	$7	$7
		Duffield, J. W., Neher, C. J. and Brown, T. C.-1992	CV			$1,256	$1,256	$1,256

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
		Duffield, J. W., Neher, C. J. and Brown, T. C.-1992	CV			$889	$889	$889
		Bowker, J. M., English, D. and Donovan, J.-1996	TC	$3,766	$9,052		$6,409	
						Aesthetic & Recreational	**$1,370**	**$608**
Riparian buffer, cont.	*Cultural & Spiritual*							
		Greenley, D., Walsh, R. G. and Young, R. A.-1981	CV			$4	$4	$4
						Cultural & Spiritual	**$4**	**$4**
						Riparian Buffer Total	**$3,382**	**$797**
Saltwater Wetland or Salt Marsh								
	Disturbance prevention							
		Farber, S.-1987	AC	$1	$1		$1	$1
		Farber, S. and Costanza, R.-1987	AC			$1	$1	$1
						Disturbance prevention	**$1**	**$1**
	Waste treatment							
		Breaux, A., Farber, S. and Day, J.-1995	AC	$1,256	$1,942		$1,599	$1,599
		Breaux, A., Farber, S. and Day, J.-1995	AC	$103	$116		$109	$109
		Breaux, A., Farber, S. and Day, J.-1995	AC			$16,560	$16,560	$16,560

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
							2004 dollars per acre/year	
						Waste treatment	**$6,090**	**$1,599**
	Refugium function & Wildlife conservation							
		Lynne, G. D., Conroy, P. and Prochaska, F. J.-1981	ME			$1	$1	$1
		Farber, S. and Costanza, R.-1987	ME			$1	$1	$1
		Bell, F. W.-1997	FI	$144	$953		$549	$549
Saltwater wetland, cont.		Batie, S. S. and Wilson, J. R.-1978	ME	$6	$735		$370	$370
						Refugium function	**$230**	**$186**
	Aesthetic & Recreational							
		Farber, S.-1988	TC	$5	$14		$9	$9
		Bergstrom, J. C., et. al. -1990	CV			$14	$14	$14
		Anderson, G. D. and Edwards, S. F.-1986	HP	$20	$91		$55	$55
						Aesthetic & Recreational	**$26**	**$14**
	Cultural & Spiritual							
		Anderson, G. D. and Edwards, S. F.-1986	CV	$120	$240		$180	$180
						Cultural & Spiritual	**$180**	**$180**
						Saltwater Wetland or Salt Marsh Total	**$6,527**	**$1,980**

Land Cover		Author(s)	Method	Min	Max	2004 dollars per acre/year Single Value	Mean	Median
Urban Green Space								
	Gas & Climate regulation							
		McPherson, E. G., Scott, K. I. and Simpson, J. R.-1998	DM			$25	$25	$25
		McPherson, E. G.-1992	AC			$820	$820	$820
		McPherson, E. G.-1992	AC			$164	$164	$164
						Gas & Climate regulation	**$336**	**$164**
Urban greenspace, cont.	*Water regulation*							
		McPherson, E. G.-1992	AC			$6	$6	$6
						Water regulation	**$6**	**$6**
	Aesthetic & Recreation							
		Tyrvainen, L.-2001	CV			$3,465	$3,465	$3,465
		Tyrvainen, L.-2001	CV			$1,182	$1,182	$1,182
		Tyrvainen, L.-2001	CV			$1,745	$1,745	$1,745
						Aesthetic & Recreation	**$2,131**	**$1,745**
						Urban Green Space Total	**$2,473**	**$1,915**

Code	SubType
DM	Direct market valuation
AC	Avoided Cost
RC	Replacement Cost
FI	Factor Income
TC	Travel Cost
HP	Hedonic Pricing
CV	Contingent Valuation
GV	Group Valuation
EA	Energy Analysis
MP	Marginal Product Estimation
CRS	Combined Revealed and Stated Preference

Appendix B

Summary of Non-Market Literature on Coastal and Nearshore Marine Systems

In this appendix, we summarize the 155 observations from the 70 studies included in the chapter. For review purposes, the observations are arranged in the fist column by land cover type and then in the second, by ecosystem service type.

In a third column, we provide the reader with a brief description of key characteristics related to each data point, including sub-service type (e.g. wildlife viewing is a sub-type of recreational service), location (specific study area), economic measures (e.g. WTP/WTA, net present value/annual value) and context change (e.g. degree of habitat loss or water quality improvement) where available. In a few cases where a median value was reported in an original study, we also add the word median into our description (see below).

In column four, citations are listed in an abbreviated form for every observation. Complete citations of all 70 peer-reviewed studies could be found at the end of the table itself. Column five features a unique code for type of valuation methodology used and the codes are listed below:

Code	**Valuation Method**
DM	Direct market valuation
AC	Avoided cost
RC	Replacement cost
TC	Travel cost
HP	Hedonic pricing
CV	Contingent valuation
TC-HP	Combined travel cost and hedonic pricing

EA	Energy analysis
MPE	Marginal product estimation
CRS	Combined revealed and stated preference

All valuation ($) estimates are documented as originally published and no conversion is applied, apart from standardizing to **2005 US Dollars** for the purpose of comparison. Annualualized conversion rates between foreign currency and US dollar were used if necessary when month-specific dates are not available from the original study.

The last four columns of the table report upper-bound, lower-bound, mean (median if noted) and the valuation unit of each observation point. The upper/lower bound and mean values correspond to statistical maximum, minimum and mean reported in the original study. If only a single midpoint estimate is reported in the original study, then it lower bound and upper bound columns are left intentionally blank.

Land Cover	Ecosystem Service	Ecosystem Services Valued	Citation	Valuation Method	Lower Bound	Upper Bound	Mean	Valuation Unit
Estuaries and Lagoons	Habitat	Saltwater marsh' s contribution to marine recreational fishing on the East coast of Florida	Bell (1997)	MPE			$1,843.98	Per acre
		Saltwater marsh' s contribution to marine recreational fishing on the West coast of Florida	Bell (1997)	MPE			$12,163.53	Per acre
	Water supply	WTP for water quality improvements from "unacceptable for swimming" to "acceptable" in Chesapeake Bay	Bockstael et al (1989)	CV	$71.43	$227.44		Per person year
		Aggregated Benefits of Improved Water Quality (safe to shell fishing) in Upper Narragansett Bay	Hayes et al (1992)	CV	$69,924,812	$133,646,616		Per year
		Aggregated Benefits of Improved Water Quality (safe to swimming) in Upper Narragansett Bay	Hayes et al (1992)	CV	$74,248,120	$115,413,533		Per year
		WTP for Preventing Eutrophication in Brest Harbor, France	Le Goffe (1995)	CV	$38.43	$39.41		Per household year
		WTP for Improved Water Quality (Safe Bathing and Shellfish Consumption) in Brest Harbor, France	Le Goffe (1995)	CV	$52.05	$52.30		Per household year
		Benefits of reducing fecal coliform counts to the state standard in Annne Arundel County, Maryland	Leggett and Bockstael (2000)	HP	$4,609,489	$24,940,389	$14,774,939	
		WTP for Water Quality and Fish Wildlife habitat in the Albemarle-Pamlico Estuarine	Whitehead et al (1995)	CV	$73.93	$106.32		Per household year

		System						
		WTP for Environmental Quality Improvement in the Pamlico Sound	Whitehead et al (1998)	CV	$280.69		$351.39	Per household year
		Consumer surplus of improved water quality in the Albemarle-Pamlico Sounds in North Carolina	Whitehead et al (2000)	CRS			$43.59	Per household season
	Recreation	Benefit loss due to loss of 35 sites with popular launch points (boat ramps) in Albemarle and Pamlico Sounds, North Carolina	Kaoru et al (1995)	TC	$5.51	$102.56		Per trip per party
		Benefit gain due to 5% increase of total catch at 35 sites with popular launch points (boat ramps) in Albemarle and Pamlico Sounds, North Carolina	Kaoru et al (1995)	TC	$11.44	$54.24		Per trip per party
		Benefit gain due to 36% decrease in nitrogen loadings at 35 sites with popular launch points (boat ramps) in Albemarle and Pamlico Sounds, North Carolina	Kaoru et al (1995)	TC	$2.13	$11.60		Per trip per party
		WTP for a larger clam fishing area in the Venice Lagoon	Nunes et al (2004)	CV	$0.29	$0.41		Per person year
		Consumer surplus of current water quality in the Albemarle and Pamlico Sounds in North Carolina	Whitehead et al (2000)	CRS			$154.53	Per household season
		Consumer surplus of improved water quality in the Albemarle and Pamlico Sounds in North Carolina	Whitehead et al (2000)	CRS			$198.13	Per household season
	Aesthetic	Stated compensating variation estimate of amenity value of the Long Island Sound	Earnhart (2001)	CV			$230,493.94	

		Revealed compensating variation estimate of amenity value of the Long Island Sound	Earnhart (2001)	HP			$8,736.49	
		Value of Lost Coastal Access Amenities for houses losing miles at Anne Arundel County, Maryland	Parsons and Wu (1991)	HP	$456.86	$1,027.45		Per house
		Value of Lost Coastal Access Amenities for houses losing view and miles at Anne Arundel County, Maryland	Parsons and Wu (1991)	HP	$12,849.02	$15,456.86		Per house
		Value of Lost Coastal Access Amenities for houses losing frontage, view and miles at Anne Arundel County, Maryland	Parsons and Wu (1991)	HP	$146,594.12	$189,552.94		Per house
Beaches and Dunes	Habitat	WTP for Preservation of Sea turtles in North Carolina	Whitehead (1993)	CV			$15.75	Per household year
	Disturbance regulation	WTP for protecting Maine and New Hampshire beaches from erosion	Lindsay et al (1992)	CV			$50.83	Per person year
		WTP for Beach Renourishment at New Jersey Beaches	Silberman and Klock (1998)	CV			$0.50	Per person day
	Water supply	Aggregated loss in use value in terms of sport fishing due to the Exxon Valdez oil spill at the upper and lower Kenai Peninsula, Anchorage, Fairbanks, Glennalen, and southeast Alaska	Hausman et al (1995)	TC	$4,080,434	$4,116,828		
		Consumer Surplus generated by Improving Water Quality of Tokyo Bay for Recreation Group 2 (includes clam-digging, paddling, and shore fishing)	Kewabe and Oka (1996)	TC			$2.561e9	Per year
		WTP for improving water quality on the Estoril Coast,	Machado and Mourato (2002)	CV	$1,435.83	$3,420.62		Per person visit

		Portugal						
		Combined CV and TC estimated benefits of Improved Water Quality of Beaches near a metropolitan area of South America	Niklitschek and Leon (1996)	CRS				Per household month
		TC estimated Benefits of Improved Water Quality of Beaches near a metropolitan area of South America	Niklitschek and Leon (1996)	TC				Per household month
		CV estimated Benefits of Improved Water Quality of Beaches near a metropolitan area of South America	Niklitschek and Leon (1996)	CV				Per household month
		Benefits of improved water quality of Beaches near a metropolitan area of South America after taking account of beach capacity	Niklitschek and Leon (1996)	CRS	$6.32	$12.73		Per household month
	Recreation	Consumer surplus of Recreation at saltwater beaches in Florida	Bell and Leeworthy (1990)	TC	$63.74	$72.29		Per person day
		Consumer surplus of recreational values in Xia Man Island, China	Chen et al (2004)	TC			$17.48	Per person trip
		Recreational Values for Beaches in South Kingston, Rhode Island	Edwards and Gable (1991)	HP			$1,111.37	Per person year
		Consumer Surplus of sport fishing at the upper and lower Kenai Peninsula, Anchorage, Fairbanks, Glennalen, and southeast Alaska	Hausman et al (1995)	TC	$187.40	$233.07		Per trip
		WTP for a recreational beach at the town of Eastbourne, UK	King (1995)	CV	$3.31	$4.14		Per person visit

		Welfare loss associated with closure of recreational saltwater fishing sites in California	Kling and Herriges (1995)	TC	$13.23	$26.06		Per fishing site choice occasion
		WTP for use of Taean-Haean National Parks in Korean	Lee and Han (2002)	CV			$5.63	Per person
		Compensating Variation for shore fishing at Clatsop County, Oregon	Marey et al (1991)	TC	$12.65	$240.04		Per person year
		Recreational welfare loss if the beach area of Zandvoort, Netherlands is closed for the entire year	Nunes and Van den Bergh (2004)	TC			$57.37	Per person year
		WTP for recreation at New Jersey beaches without renourishment	Silberman and Klock (1998)	CV			$5.94	Per person day
		WTP for recreation at New Jersey beaches with renourishment	Silberman and Klock (1998)	CV			$6.44	Per person day
	Aesthetic	Median WTP for different beach debris conditions in North Carolina	Smith et al (1997)	CV	$29.78 (Median)	$100.53 (Median)		Per household year
	Spiritual and historic	One time WTP for existence of New Jersey beaches with renourishment	Silberman and Klock (1998)	CV			$26.91	Per person
		Non-users' existence value for New Jersey beaches in the form of one-time WTP based on telephone survey	Silberman et al(1992)	CV			$13.25	Per person
		Non-users' existence value for New Jersey beaches in the form of one-time WTP based on in-site survey	Silberman et al(1992)	CV			$13.01	Per person
Salt-water Wetland, Marsh or Salt-Pond	Habitat	One time WTP for more species at North Berwick, Scotland	Edwards-Jones et al (1995)	CV	$7.59	$7.88		Per person

		One time WTP for more species at Yellowcraigs, Scotland	Edwards-Jones et al (1995)	CV	$8.06	$8.90		Per person
		Marginal value of marsh for blue crab fishery on Florida's Gulf Coast	Lynne et al (1981)	MPE			$1.09	Per acre
	Disturbance regulation	Present value of coastal Louisiana wetland in providing storm protection services	Costanza et al (1989)	RC	$3,754.90	$14,801.96		Per acre
		Hurricane damages due to a coastal recession of one Mile of wetlands at the Gulf Coast of Mexico, Louisiana	Farber (1987)	RC	$0.01	$0.38		Per person year
	Water supply	Local residents' net present value to prevent water quality deterioration for coastal salt ponds in Rhode Island	Anderson and Edwards (1986)	CV			$294.12	Per person
		Median WTP for avoiding damage to the Coorong due to drainage of saline water from surrounding agricultural areas into the wetlands	Bennett et al (1998)	CV			$68.00 (Median)	Per household
		WTP for improving water quality to allow year-round shell fishing at three coastal ponds in Martha's Vineyard Island, Massachusetts	Kaoru (1993)	CV			$177.07	Per household year
		One time WTP for restoration of a historic salt marsh, West River Memorial Park, Connecticut	Udziela and Bennett (1997)	CV			$76.48	Per household
	Recreation	Net present value of water view for houses with water frontage in Rhode Island	Anderson and Edwards (1986)	HP	$8,382.35	$39,215.69		
		Consumer surplus of wetlands-based recreation, Louisiana	Bergstrom et al (1990)	CV			$641.71	Per person year
		Present value of Louisiana coastal wetland in providing recreational services	Costanza et al (1989)	CRS	$90.20	$354.90		Per acre

		WTP for recreation at Yellowcraigs, Scotland	Edwards-Jones et al (1995)	CV	$23.66	$32.56		Per person
		WTP for recreation at North Berwick, Scotland	Edwards-Jones et al (1995)	CV	$23.40	$30.99		Per person
		WTP for preserving coastal wetlands in Terrebonne Parish, Louisiana	Farber (1988)	TC			$170.76	Per household year
		Aggregated WTP for recreation at coastal wetlands in Terrebonne Parish, Louisiana.	Farber (1988)	TC	$6,432,343	$11,887,788		Per year
		Use value of improving water quality to allow year-round shell-fishing at three coastal ponds in Martha's Vineyard Island, Massachusetts	Kaoru (1993)	CV			$45.53	Per household year
		Option value of improving water quality to allow year-round shell-fishing at three coastal ponds in Martha's Vineyard Island, Massachusetts	Kaoru (1993)	CV			$26.23	Per household year
	Aesthetic	Stated compensating variation of amenity value for restoring Pine Creek Marsh, Fairfield Connecticut	Earnhart (2001)	CV			$232,140.02	
		Revealed compensating variation of amenity value for the restoring Pine Creek Marsh, Fairfield Connecticut	Earnhart (2001)	HP			$44,738.70	
		Amenity value provided by 5 acre marsh on Virginia Beach	Shabman and Bertelson (1979)	HP			$229,548.39	
	Spiritual and historic	Existence value for improving water quality to allow year-round shell-fishing at three coastal ponds in Martha's Vineyard Island, Massachusetts	Kaoru (1993)	CV			$104.85	Per household year

	Net primary production	Present value of Louisiana coastal wetland based on Energy Analysis	Costanza et al (1989)	EA	$12,549.02	$55,294.12		Per acre
near-shore Fresh-water Wetland	Disturbance regulation	Median WTP for beach erosion management through nourishment at Jekyll Island, Georgia	Kriesel et al (2004)	CV			$7.26 (Median)	Per household day
		Present value of wetlands wastewater treatment (potato chip manufacturing waste) at Grammercy, Louisiana	Breaux et al (1995)	RC			$62,976.41	Per acre
		Present value of wetlands wastewater treatment (municipal wastewater effluent) at Thibodaux, Louisiana	Breaux et al (1995)	RC	$1,424.68	$4,174.23		Per acre
		Present value of wetlands wastewater treatment (Seafood processing Waste) at Dulac, Louisiana	Breaux et al (1995)	RC	$11,308.53	$17,486.39		Per acre
	Water supply	Median WTP for avoiding damage to Tilley Swamp result from drainage of saline water from surrounding agricultural areas into the wetlands	Bennett et al (1998)	CV			$126.28 (Median)	Per household
	Recreation	Compensating variation for an access to fishing sites in nine counties of Florida	Bockstael et al (1989)	TC	$1.28	$12.50		Per person choice occasion
		Consumer Surplus of pleasure boating at the upper and lower Kenai Peninsula, Anchorage, Fairbanks, Glennalen, and southeast Alaska	Hausman et al (1995)	TC	$357.48	$485.04		Per trip
		WTP in the form of an additional user fee for recreation at Clay Marshes Nature Reserve, England	Klein and Bateman (1998)	CV			$3.06	Per person
		WTP in the form of annual tax for recreation at Clay Marshes	Klein and Bateman	CV			$93.54	Per household

		Nature Reserve, England	(1998)					year
		TC estimated WTP for Recreation at Clay Marshes Nature Reserve, England	Klein and Bateman (1998)	TC			$88.43	Per visiting party year
	Aesthetic	Amenity benefits of coastal farm land in Suffolk County, NY	Johnston et al (2001)	CV			$0.09	Per household acre year
Sea-grass beds or Kelp forest	Habitat	WTP for protecting Florida Manatee	Solomon et al (2004)	AC			$16.27	Per household year
	Biological regulation	Aquatic vegetation removal service provided by Florida manatee	Solomon et al (2004)	AC			$33,076.07	Per year
	Aesthetic	Amenity benefits of coastal farm land in Suffolk County, NY	Johnston et al (2001)	CV			$0.13	Per household acre year
Near-shore Islands	Habitat	Open ended CV estimated WTP for continuously conserving Little Barrier Island, Auckland, New Zealand	Mortimer et al (1996)	CV			$38.68	Per household year
		Dichotomous Choice CV estimated WTP for continuously conserving Little Barrier Island, Auckland, New Zealand	Mortimer et al (1996)	CV	$29.88	$79.13	$46.46	Per household year
	Disturbance regulation	Median WTP for managing beach erosion through retreat at Jekyll Island, Georgia	Kriesel et al (2004)	CV			$9.23 (Median)	Per household day
	Recreation	Net WTP for Recreational Fishing in the Lower Atchafalaya River Basin, Louisiana	Bergstrom et al (2004)	TC-HP	$578.48	$1,111.61		Per person year
Coral Reefs and Atolls	Water supply	Change in consumer surplus for Improving water quality by 100% over the five-year study period in the Florida Keys	Bhat (2003)	CRS			$3,727.27	Per person 5 years

	Recreation	Visitors' daily WTP for entering a Philippine Marine Sanctuary	Arin and Kramer (2002)	CV	$3.69	$5.97		Per person day
		Change in consumer surplus for increasing fish abundance by 200% over the five-year study period in the Florida Keys	Bhat (2003)	CRS			$2,875.47	Per person 5 years
		Change in consumer surplus for improving coral quality by 100% over the five-year study period in the Florida Keys	Bhat (2003)	CRS			$3,835.62	Per person 5 years
		Consumer surplus per person under current Coral Reef Quality in the Florida Key over the five-year study period	Bhat (2003)	CRS			$3,641.34	Per person 5 years
		Aggregated consumer surplus for recreation at the Great Barrier Reef, Australia	Carr and Mendelsohn (2003)	TC	$753,715,499	$1,698,513,800		Per year
		WTP for snorkeling trips to Florida Keys	Park et al (2002)	CV			$510.75	Per person year
		Use value of snorkeling trips to Florida Keys	Park et al (2002)	TC			$337.13	Per person year
		Tourists' WTP in the form of an entry fee for preserving the Pulau Payar Marine Park, Malaysia	Yeo (2002)	CV	$4.11	$8.54		Per person
Man-grove	Habitat	Loss in revenue of shrimp production due to mangrove deforestation in Capmeche State, Mexico	Barbier and Strand (1998)	MPE			$388,167.13	Per year
Semi-enclosed Sea	Habitat	Median WTP for transplanting 10 hectare of eel grass (Zostera) in Seto Inland Sea, Japan	Tsuge and Washida (2003)	CV	$39.63 (Median)	$46.91(Median)		Per household
		Median WTP for protecting habitat of rare animal species in Seto Inland Sea, Japan	Tsuge and Washida (2003)	CV	$68.95 (Median)	$83.71(Median)		Per household
	Aesthetic	Median WTP to restoring four hectare shoreland in Seto Inland Sea, Japan	Tsuge and Washida (2003)	CV	$38.44 (Median)	$54.72(Median)		Per household

Near-shore Ocean--50m depth or 100km offshore	Habitat	WTP for a Doubling in the Current Salmon and Striped Bass Catch Rate in the San Francisco Bay and Ocean Area	Cameron and Huppert (1989)	CV	$102.42	$106.19		Per person year
		US People's WTP for an expanded federal protection program for the Steller Sea Lion (Eumetopias jubatus).	Giraud et al (2002)	CV			$108.82	Per person year
		People at Coastal Boroughs of Alaska's WTP for an expanded federal protection program for the Steller Sea Lion (Eumetopias jubatus).	Giraud et al (2002)	CV			-$276.58	Per person year
		People of Alaska State's WTP for an expanded federal protection program for the Steller Sea Lion (Eumetopias jubatus).	Giraud et al (2002)	CV			$43.88	Per person year
	Water supply	Consumer surplus loss to Montauk charter boat anglers of striped bass recreational fishing due to water deterioration in Chesapeake Bay	Kahn and Buerger (1994)	TC	$194.19	$506.35		Per person year
		Consumer surplus generated by improving water quality of Tokyo Bay for Recreation Group 3 (bathing, snorkeling, and surfing)	Kawabe and Oka (1996)	TC			$1.37e8	Per year
		WTP for preventing impacts caused by harmful algal bloom species (HABs) along the coastline of the Netherlands	Nunes and Van den Bergh (2004)	CV	$52.27	$79.48		Per person year
		British Columbia residents' WTP for preventing oil spills in	Rowe et al (1992)	CV	$50.20	$210.16		Per household

		the Pacific Northwest over five years						
		Washington State residents' WTP for preventing oil spills in the Pacific Northwest over five years	Rowe et al (1992)	CV	$39.27	$230.00		Per household
	Recreation	Consumer surplus per bluefish caught by anglers in states along Atlantic Coast from New York to Florida	Agnello (1989)	TC	$0.74	$1.91		Per fish
		Consumer surplus per flounder caught by anglers in states along Atlantic Coast from New York to Florida	Agnello (1989)	TC	$3.54	$15.88		Per fish
		Consumer surplus per weakfish caught by anglers in states along Atlantic Coast from New York to Florida	Agnello (1989)	TC	$0.05	$3.08		Per fish
		WTP for an extra Chinook salmon catch on the south coast of the British Columbia, Canada	Cameron and James (1987)	CV			$24.86	Per fish
		WTP loss for recreational saltwater fishing in Coastal Texas due to a 10% reduction in fishing days	Cameron (1992)	CRS	$32.65	$89.35	$60.14	Per person year
		WTP loss for Recreational Saltwater Fishing in Coastal Texas due to a 100% reduction in fishing days	Cameron (1992)	CRS	$3,190.72	$5,929.55	$8,817.87	Per person year
		Marginal increase in consumer surplus for an additional Threadfin catch in Hawaii	Cantrell et al (2004)	CV			$2.50	Per fish
		Median Willingness to Pay for recreational saltwater fishing in Galveston, Texas Bay area	Downing and Ozuna (1996)	CV	$127.43 (Median)	$406.61(Median)		Per person year
		Median Willingness to Pay for recreational saltwater fishing in Lower Laguna Madre, Texas Bay area	Downing and Ozuna (1996)	CV	$155.12 (Median)	$244.02(Median)		Per person year

		Median Willingness to Pay for recreational saltwater fishing in San Antonio, Texas Bay area	Downing and Ozuna (1996)	CV	$125.43 (Median)	$162.39(Median)		Per person year
		Median Willingness to Pay for recreational saltwater fishing in Aransas, Texas Bay area	Downing and Ozuna (1996)	CV	$187.75 (Median)	$240.47(Median)		Per person year
		Median Willingness to Pay for recreational saltwater fishing in Sabine, Texas Bay area	Downing and Ozuna (1996)	CV	$60.03 (Median)	$133.50(Median)		Per person year
		Median Willingness to Pay for recreational saltwater fishing in Corpus Christi, Texas Bay area	Downing and Ozuna (1996)	CV	$133.89 (Median)	$191.83(Median)		Per person year
		Median Willingness to Pay for recreational saltwater fishing in Upper Laguna Madre, Texas Bay area	Downing and Ozuna (1996)	CV	$130.80 (Median)	$205.12(Median)		Per person year
		Median Willingness to Pay for recreational saltwater fishing in Matagorda, Texas Bay area	Downing and Ozuna (1996)	CV	$71.18 (Median)	$186.98(Median)		Per person year
		Actual expenditures made by Killer Whales watchers in Johnstone Strait off British Columbia's Vancouver Island	Duffus and Dearden (1993)	DM	$490.03	$529.13		Per person trip
		Increased consumer surplus due to a 100% increase in salmon and striped bass catch in San Francisco Bay area	Huppert (1989)	TC	$96.06	$466.14		Per person trip
		WTP for a 100% increase in salmon and striped bass catch in San Francisco Bay area	Huppert (1989)	CV			$77.48	Per person year
		Welfare loss due to closure of all offshore recreational saltwater fishing sites in California	Kling and Herriges (1995)	TC	$43.24	$70.00		per fishing site choice occasion
		WTP for use of Hallyo-Haesang National Parks in Korean	Lee and Han (2002)	CV			$15.36	Per person

		Net present value loss of ocean sport salmon fishing due to timber harvesting in the Siuslaw National Forest, Oregon	Loomis (1988)	TC			$968,646.86	
		Net present value of ocean sport salmon fishing under the influence of forest management practice of the Siuslaw National Forest, Oregon	Loomis (1988)	TC	$1,392,739.2 7	$2,361,386.14		
		Compensating variation for boat fishing at Clatsop County, Oregon	Morey et al (1991)	TC	$6.92	$130.04		Per person year
		Economic income generated by cetacean-related tourism in rural West Scotland	Parsons et al (2003)	DM	$3.05e8	$8.789e8		Per year
	Aesthetic	Amenity benefits of coastal farm land in in Suffolk County, NY	Johnston et al (2001)	CV			$0.08	Per household acre year
Near-shore Open Space	Habitat	WTP for the Wilderness Area Programs in the Parque Natural Alentejano e Costa Vicentina, Portugal	Nunes (2002)	CV	$48.91	$106.57		Per household year
	Water supply	Compensating variation for the elimination of La Victoria recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$0.15	Per person day
		Aggregated loss in use value in terms of hunting due to the Exxon Valdez oil spill at the upper and lower Kenai Peninsula, Anchorage, Fairbanks, Glennalen, and southeast Alaska	Hausman et al (1995)	TC	$340,519.69	$546,066.14		
		Aggregated loss in use value in terms of hiking/viewing due to the Exxon Valdez oil spill at the upper and lower Kenai	Hausman et al (1995)	TC	$393,267.72	$1,720,023.62		

		Peninsula, Anchorage, Fairbanks, Glennalen, and southeast Alaska						
		Consumer surplus generated by improving water quality of Tokyo Bay for recreation group 1 (includes viewing, walking, nature study, photography, and sketching)	Kewabe and Oka (1996)	TC			$2,875,000,000.00	Per year
	Recreation	Compensating variation per individual per day for the elimination of Sa Calbora recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$0.90	Per person day
		Compensating variation per individual per day for the elimination of Es Trenc-Salobrar de Campos recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$1.03	Per person day
		Compensating variation per individual per day for the elimination of Mondrago recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$0.10	Per person day
		Compensating variation per individual per day for the elimination of Formentor recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$1.97	Per person day
		Compensating variation per individual per day for the elimination of Cala Agulla recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$0.75	Per person day
		Compensating variation per individual per day for the	Font (2000)	TC			$0.09	Per person day

		elimination of Cala Figuera recreational site, Mallorca, the Balearic Island						
		Compensating variation per individual per day for the elimination of Ca de Ses Salines recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$0.06	Per person day
		Compensating variation per individual per day for the elimination of S'Albufera recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$0.19	Per person day
		Compensating variation per individual per day for the elimination of Punta de n' Amer recreational site, Mallorca, the Balearic Island	Font (2000)	TC			$0.05	Per person day
		Consumer surplus for hunting at the upper and lower Kenai Peninsula, Anchorage, Fairbanks, Glennalen, and southeast Alaska	Hausman et al (1995)	TC	$77.17	$633.07		Per trip
		Consumer surplus for hiking/viewing at the upper and lower Kenai Peninsula, Anchorage, Fairbanks, Glennalen, and southeast Alaska	Hausman et al (1995)	TC	$305.51	$612.60		Per trip
		WTP values for use of Soraksan National Parks in Korean	Lee and Han (2002)	CV			$16.76	Per person

		WTP for the Recreation Area Programs in the Parque Natural Portugal	Nunes (2002)	CV	$37.96	$85.40		Per household year
	Aesthetic	Amenity benefits of coastal farm land in Suffolk County, NY	Johnston et al (2001)	CV			$0.04	Per household acre year
		Amenity benefits of coastal farm land in Suffolk County, NY	Johnston et al (2001)	CV			$0.16	Per household acre year

Marine and Coastal Non-Market Study References

1. Agnello, R. J. 1989. "The Economic Value Of Fishing Success - An Application Of Socioeconomic Survey Data." *Fishery Bulletin* 87:223-232.
2. Anderson, G. D. and S. F. Edwards. 1986. "Protecting Rhode-Island Coastal Salt Ponds - an Economic-Assessment of Downzoning." *Coastal Zone Management Journal* 14:67-91.
3. Arin, T. and R. A. Kramer. 2002. "Divers' willingness to pay to visit marine sanctuaries: an exploratory study." *Ocean & Coastal Management* 45:171-183.
4. Barbier, E. B. and I. Strand. 1998. "Valuing mangrove-fishery linkages. A case study of Campeche, Mexico." *Environmental and Resource Economics* 12:151-166.
5. Bell, F. W. 1997. "The economic valuation of saltwater marsh supporting marine recreational fishing in the southeastern United States." *Ecological Economics* 21:243-254.
6. Bell, F. W. and V. R. Leeworthy. 1990. "Recreational Demand by Tourists for Saltwater Beach Days." *Journal of Environmental Economics and Management* 18:189-205.
7. Bennett, J., M. Morrison, and R. Blamey. 1998. "Testing the validity of responses to contingent valuation questioning." *Australian Journal Of Agricultural And Resource Economics* 42:131-148.
8. Bergstrom, J. C., J. H. Dorfman, and J. B. Loomis. 2004. "Estuary management and recreational fishing benefits." *Coastal Management* 32:417-432.

9. Bergstrom, J. C., J. R. Stoll, J. P. Titre, and V. L. Wright. 1990. "Economic value of wetlands-based recreation." *Ecological Economics* 2:129-147.

10. Bhat, M. G. 2003. "Application of non-market valuation to the Florida Keys marine reserve management." *Journal of Environmental Management* 67:315-325.

11. Bockstael, N. E., K. E. McConnell, and I. Strand. 1989. "A random utlity model for sportfishing: some preliminary results for Florida." *Marine Resource Economics* 6:245-260.

12. Bockstael, N.E., K.E. McConnell, and I.E. Strand. 1989. "Measuring the Benefits of Improvements in Water Quality: The Chesapeake Bay." *Marine Resource Economics* 6:1-18.

13. Breaux, A., S. Farber, and J. Day. 1995. "Using Natural Coastal Wetlands Systems for Waste-Water Treatment - an Economic Benefit Analysis." *Journal of Environmental Management* 44:285-291.

14. Cameron, T. A. 1992. "Combining Contingent Valuation and Travel Cost Data for the Valuation of Nonmarket Goods." *Land Economics* 68:302-317.

15. Cameron, T. A. and D. D. Huppert. 1989. "Ols Versus Ml Estimation of Non-Market Resource Values with Payment Card Interval Data." *Journal of Environmental Economics and Management* 17:230-246.

16. Cameron, T. A. and M. D. James. 1987. "Efficient Estimation Methods for Closed-Ended Contingent Valuation Surveys." *Review of Economics and Statistics* 69:269-276.

17. Cantrell, R. N., M. Garcia, P. S. Leung, and D. Ziemann. 2004. "Recreational

anglers' willingness to pay for increased catch rates of Pacific threadfin (Polydactylus sexfilis) in Hawaii." *Fisheries Research* 68:149-158.

18. Carr, L. and R. Mendelsohn. 2003. "Valuing coral reefs: A travel cost analysis of the Great Barrier Reef." *Ambio* 32:353-357.
19. Chen, W. Q., H. S. Hong, Y. Liu, L. P. Zhang, X. F. Hou, and M. Raymond. 2004. "Recreation demand and economic value: An application of travel cost method for Xiamen Island." *China Economic Review* 15:398-406.
20. Costanza, R., S. C. Farber, and J. Maxwell. 1989. "Valuation and management of wetland ecosystems." *Ecological Economics* 1:335-361.
21. Downing, M. and T. Ozuna. 1996. "Testing the reliability of the benefit function transfer approach." *Journal Of Environmental Economics And Management* 30:316-322.
22. Duffus, D. A. and P. Dearden. 1993. "Recreational Use, Valuation, And Management, Of Killer Whales (Orcinus-Orca) On Canada Pacific Coast." *Environmental Conservation* 20:149-156.
23. Earnhart, D. 2001. "Combining revealed and stated preference methods to value environmental amenities at residential locations." *Land Economics* 77:12-29.
24. Edwards, S. F. and F.J. Gable. 1991. "Estimating the value of beach recreation from property values: an exploration with comparisons to nourishment costs." *Ocean & Shoreline Management* 15:37-55.
25. Edwards-Jones, G., E.S. Edwards-Jones, and K. Mitchell. 1995. "A Comparison of Contingent Valuation Methodology and Ecological Assessment as Techniques for Incoporating Ecological Goods into Land-use Decisions." *Journal of*

Environmental Planning and Management 38:215-230.

26. Farber, S. 1987. "The Value of Coastal Wetlands for Protection of Property against Hurricane Wind Damage." *Journal of Environmental Economics and Management* 14:143-151.
27. —. 1988. "The Value of Coastal Wetlands for Recreation - an Application of Travel Cost and Contingent Valuation Methodologies." *Journal of Environmental Management* 26:299-312.
28. Font, A. R. 2000. "Mass tourism and the demand for protected natural areas: A travel cost approach." *Journal Of Environmental Economics And Management* 39:97-116.
29. Giraud, K., B. Turcin, J. Loomis, and J. Cooper. 2002. "Economic benefit of the protection program for the Steller sea lion." *Marine Policy* 26:451-458.
30. Hausman, J. A., G. K. Leonard, and D. McFadden. 1995. "A Utility-Consistent, Combined Discrete-Choice and Count Data Model Assessing Recreational Use Losses Due to Natural-Resource Damage." *Journal of Public Economics* 56:1-30.
31. Hayes, K.M., T.J. Tyrrell, and G. Anderson. 1992. "Estimating the benefits of water quality improvements in the Upper Narragansett Bay." *Marine Resource Economics* 7:75-85.
32. Huppert, D. D. 1989. "Measuring the value of fish to anglers: application to central California Anadromous species." *Marine Resource Economics* 6:89-107.
33. Johnston, R. J., J. J. Opaluch, T. A. Grigalunas, and M. J. Mazzotta. 2001. "Estimating amenity benefits of coastal farmland." *Growth And Change* 32:305-325.

34. Kahn, J. R. and R. B. Buerger. 1994. "Valuation and the Consequences of Multiple Sources of Environmental Deterioration - the Case of the New-York Striped Bass Fishery." *Journal of Environmental Management* 40:257-273.

35. Kaoru, Y. 1993. "Differentiating Use and Nonuse Values for Coastal Pond Water Quality Improvements." *Environmental & Resource Economics* 3:487-494.

36. Kaoru, Y., V. K. Smith, and J. L. Liu. 1995. "Using Random Utility-Models to Estimate the Recreational Value of Estuarine Resources." *American Journal of Agricultural Economics* 77:141-151.

37. Kawabe, M. and T. Oka. 1996. "Benefit from improvement of organic contamination of Tokyo Bay." *Marine Pollution Bulletin* 32:788-793.

38. King, Oliver H. 1995. "Estimating the value of marine resources: a marine recreation case." *Ocean & Coastal Management* 27:129-141.

39. Klein, R. J. T. and I. J. Bateman. 1998. "The recreational value of Cley Marshes nature reserve: An argument against managed retreat?" *Journal Of The Chartered Institution Of Water And Environmental Management* 12:280-285.

40. Kling, C. L. and J. A. Herriges. 1995. "An Empirical-Investigation of the Consistency of Nested Logit-Models with Utility Maximization." *American Journal of Agricultural Economics* 77:875-884.

41. Kriesel, W., A. Keeler, and C. Landry. 2004. "Financing beach improvements: Comparing two approaches on the Georgia coast." *Coastal Management* 32:433-447.

42. Le Goffe, Phillippe. 1995. "The Benefits of Improvements in Coastal Water Quality: A Contingent Approach." *Journal of Environmental Management*

45:305-317.

43. Lee, C. K. and S. Y. Han. 2002. "Estimating the use and preservation values of national parks' tourism resources using a contingent valuation method." *Tourism Management* 23:531-540.

44. Leggett, C. G. and N. E. Bockstael. 2000. "Evidence of the effects of water quality on residential land prices." *Journal of Environmental Economics and Management* 39:121-144.

45. Lindsay, B. E., J. M. Halstead, H. C. Tupper, and J. J. Vaske. 1992. "Factors Influencing the Willingness to Pay for Coastal Beach Protection." *Coastal Management* 20:291-302.

46. Loomis, J. B. 1988. "The bioeconomic effects of timber harvesting on recreational and commercial salmon and steelhead fishing: a case study of the Siuslaw National Forest." *Marine Pollution Bulletin* 5:43-60.

47. Lynne, G.D., P. Conroy, and F.J. Prochaska. 1981. "Economic Valuation of Marsh Areas for Marine Production Processes." *Journal of Environmental Economics and Management* 8:175-186.

48. Machado, F. S. and S. Mourato. 2002. "Evaluating the multiple benefits of marine water quality improvements: how important are health risk reductions?" *Journal Of Environmental Management* 65:239-250.

49. Morey, E. R., W. D. Shaw, and R. D. Rowe. 1991. "A Discrete-Choice Model of Recreational Participation, Site Choice, and Activity Valuation When Complete Trip Data Are Not Available." *Journal of Environmental Economics and Management* 20:181-201.

50. Mortimer, R., B. Sharp, and J. Craig. 1996. "Assessing the conservation value of New Zealand's offshore islands." *Conservation Biology* 10:25-29.

51. Niklitschek, M. and J. Leon. 1996. "Combining intended demand and yes/no responses in the estimation of contingent valuation models." *Journal Of Environmental Economics And Management* 31:387-402.

52. Nunes, Pald. 2002. "Using factor analysis to identify consumer preferences for the protection of a natural area in Portugal." *European Journal Of Operational Research* 140:499-516.

53. Nunes, Pald, L. Rossetto, and A. de Blaeij. 2004. "Measuring the economic value of alternative clam fishing management practices in the Venice Lagoon: Results from a conjoint valuation application." *Journal Of Marine Systems* 51:309-320.

54. Nunes, Pald and Jcjm Van den Bergh. 2004. "Can people value protection against invasive marine species? Evidence from a joint TC-CV survey in the Netherlands." *Environmental & Resource Economics* 28:517-532.

55. Park, T., J. M. Bowker, and V. R. Leeworthy. 2002. "Valuing snorkeling visits to the Florida Keys with stated and revealed preference models." *Journal of Environmental Management* 65:301-312.

56. Parsons, E. C. M., C. A. Warburton, A. Woods-Ballard, A. Hughes, and P. Johnston. 2003. "The value of conserving whales: the impacts of cetacean-related tourism on the economy of rural West Scotland." *Aquatic Conservation-Marine And Freshwater Ecosystems* 13:397-415.

57. Parsons, G. R. and Y. R. Wu. 1991. "The Opportunity Cost Of Coastal Land-Use Controls - An Empirical-Analysis." *Land Economics* 67:308-316.

58. Rowe, R.D., W.D. Shaw, and W. Schulze. 1992. "Nestucca Oil Spill." in *Natural Resource Damages*, edited by K. Ward and J. Duffield. New York: Wiley and Sons.
59. Shabman, L. and M. K. Bertelson. 1979. "The use of development value estimates for coastal wetland permit decisions." *Land Economics* 55:213-222.
60. Silberman, J., D. A. Gerlowski, and N. A. Williams. 1992. "Estimating Existence Value for Users and Nonusers of New-Jersey Beaches." *Land Economics* 68:225-236.
61. Silberman, Jonathan and Mark Klock. 1988. "The Recreation Benefits of Beach Renourishment." *Ocean & Shoreline Management* 11:73-90.
62. Smith, V.K., X. Zhang, and R.B. Palmquist. 1997. "Marine debris, beach quality, and non-market values." *Environmental and Resource Economics* 10:223-247.
63. Solomon, B. D., C. M. Corey-Luse, and K. E. Halvorsen. 2004. "The Florida manatee and eco-tourism: toward a safe minimum standard." *Ecological Economics* 50:101-115.
64. Tsuge, T. and T. Washida. 2003. "Economic valuation of the Seto Inland Sea by using an Internet CV survey." *Marine Pollution Bulletin* 47:230-236.
65. Udziela, M.K. and L.L. Bennett. 1997. "Contingent valuation of an urban salt marsh restoration." *Yale Forestry and Environmental Studies Bulletin* 100:41-61.
66. Whitehead, J.C. 1993. "Total economic values for coastal and marine wildlife: specification, validity, and valuation issues." *Marine Resource Economics* 8.
67. Whitehead, J. C., G. C. Blomquist, T. J. Hoban, and W. B. Clifford. 1995. "Assessing the Validity and Reliability of Contingent Values - a Comparison of

on-Site Users, Off-Site Users, and Nonusers." *Journal of Environmental Economics and Management* 29:238-251.

68. Whitehead, J. C., G. C. Blomquist, R. C. Ready, and J. C. Huang. 1998. "Construct validity of dichotomous and polychotomous choice contingent valuation questions." *Environmental & Resource Economics* 11:107-116.
69. Whitehead, J. C., T. C. Haab, and J. C. Huang. 2000. "Measuring recreation benefits of quality improvements with revealed and stated behavior data." *Resource and Energy Economics* 22:339-354.
70. Yeo, B.H. 2002. "Valuing a Marine Park in Malaysia." Pp. 311-324 in *Valuing the Environment in Developing Countries: Case Studies,*, edited by D. Pearce, C. Pearce, and C. Palmer. Cheltenham, UK,Northampton, MA, USA: Edward Elgar.