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Robert Costanza
Portland State University

Matthew A. Wilson

Austin Troy

Alexey Voinov

Shang Liu

See next page for additional authors

Authors
Robert Costanza, Matthew A. Wilson, Austin Troy, Alexey Voinov, Shang Liu, and John D'Agostino

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Robert Costanza
Matthew Wilson
Austin Troy
Alexey Voinov
Shuang Liu
John D'Agostino

Gund Institute for Ecological Economics
Rubenstein School of Environment and Natural Resources
University of Vermont
Burlington, VT 05405

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Executive Summary

This report summarizes the results of a two-year study of the economic value of New Jersey's natural capital. Natural capital consists of those components of the natural environment that provide aa long-term stream of benefits to individual people and to society as a whole; the value of natural capital is defined in this report as the present value of that benefit stream. Many of the benefits provided by natural capital come from ecological systems ("ecosystems"); an ecosystem is a dynamic complex of plant, animal, and microorganism communities and their nonliving environment, all interacting as a functional unit.

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 are mainly provided by ecosystems. Examples of ecosystem services ("ecoservices") include temporary storage of flood waters 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 human beings. The goods provided by New Jersey's natural capital are covered in a separate study; this report focuses on the services provided the state's ecosystems, covering twelve different types of ecosystem and twelve different ecoservices.

For policy, planning, and regulatory decisions, it is important for New Jerseyans 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 marketed and non-marketed goods and services, such as those provided by physical capital (e.g., roads), human capital investment (e.g., education), etc. As a way of expressing these relative values or "trade-offs", this study estimated the dollar value of the ecoservices produced by New Jersey's ecosystems. In deriving these estimates, we used three different approaches: value transfer, hedonic analysis, and spatial modeling.

A. Value Transfer

Value transfer identifies previously conducted high-quality studies of the value of ecoservices in a variety of locations using a variety of valuation methods and applies them to New Jersey ecosystems. Value transfer is the preferred valuation technique where (as in this case) performing original research for an extended geographic region with varied ecosystem types would be prohibitively expensive.

For the present study, we identified and used a total of 100 earlier studies covering the types of ecosystems present in New Jersey; 94 of these studies are original research previously published in peer-reviewed journals. Some studies provided more than one estimated ecoservice value for a given ecosystem; the set of 100 studies provided a total of 210 individual value estimates. We translated each estimate into dollars per acre per year, computed the average value for a given ecoservice for a given ecosystem, and multiplied the average by the total statewide acreage for that ecosystem.

Our results are summarized below; all figures are 2004 dollars. The figures include only ecosystem services; they do not include ecosystem or abiotic goods or secondary economic activity related to a given ecosystem.

1. Wetlands provided the largest dollar value of ecosystem services: $9.4 billion/yr for freshwater wetlands and $1.2 billion/yr for saltwater wetlands. The most valuable services were disturbance regulation ($3.0 billion/yr), water filtration ($2.4 billion/yr), and water supply ($1.3 billion/yr) for freshwater wetlands, and waste treatment ($1.0 billion/yr) for saltwater wetlands. (Disturbance regulation means the buffering of floods, storm surges, and other events that threaten things valued by individuals or by society as a whole.)

2. Marine ecosystems provided the second-largest dollar amount of ecosystem services: $5.3

billion/yr for estuaries and tidal bays and about $389 million/yr for other coastal waters, including the coastal shelf out to the three-mile limit. (It should be noted that the fish and shellfish obtained from these ecosystems are covered elsewhere in this report and are not included in these totals.) Nutrient cycling (i.e., waste dilution and removal) was the most important service provided by marine ecosystems, with a value of $5.1 billion/yr.

3. Forests cover the largest area of any ecosystem type in New Jersey, and because of that the total value of the ecosystem services they provide is one of the highest at $2.2 billion/yr, excluding the value of timber. Habitat services are currently the most important of these services ($1.4 billion/yr); other important services provided by forests include water supply and pollination (about $238 million/yr each) and aesthetic and recreational amenities ($179 million/yr).

4. Urban green space covers relatively little of New Jersey but has a relatively high dollar value per acre and provides an estimated $419 million of ecosystem services annually, principally aesthetic and recreational amenities ($361 million/yr). Ecoservice values for other types of urban land and for barren land were not investigated in this study.

5. Beaches (including dunes) provided by far the highest ecoservice value per acre; their small area limited their annual ecoservice value to about $330 million, mainly disturbance regulation ($214 million/yr) and aesthetic and recreational amenities ($116 million/yr).

6. Agricultural land includes both cropland (estimated at $78 million/yr of ecosystem services) and pastureland (estimated at $45 million/yr). These values relate solely to the services provided by farmland, mainly habitat services from cropland ($75 million/yr) and waste treatment services from pasture land ($26 million/yr). They do not include the value of the food provided by farms, which is covered elsewhere.

7. Open fresh water and riparian buffers provided services with an estimated annual value of $66 million and $51 million respectively, mainly water supply ($64 million/yr) and aesthetic and recreational amenities ($51 million/yr). Another part of this report covers the value of water as an ecosystem good.

The total value of these ecosystem services is $19.4 billion/year. If we exclude studies which were not peer-reviewed and/or which did not report on original research, the result is a lower estimate of $11.6 billion/year. However, this exclusion makes it impossible to estimate values for a number of ecosystems and/or ecoservices, and we believe that the higher figure better represents the value of the services provided by New Jersey's ecosystems. If the excluded studies are added back but weighted at 50%, the total value of ecosystem services would be $15.5 billion/year.

Future flows of ecoservices 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 $19.4 billion/yr of ecoservices continues in perpetuity, the present value of those services, i.e., the value of the natural capital which provides the services, would be $648 billion. Using the same assumptions, the present values of the $11.6 billion/yr and $15.5 billion/yr flows of services (see above) would be $387 billion and $517 billion respectively.

Many decisions on environmental policy and land use are made at the local level, and it is therefore important to translate the statewide results described above into local values. Based on the results of the value transfer analysis, we mapped the aggregate value of ecosystem services by county, by watershed, and by sub-watershed. The maps show substantial differences in ecoservice values based on the predominant types of land cover in different parts of the state. In general, areas containing wetlands, estuaries, tidal bays, and beaches had the highest ecosystem service values per acre. Our maps are based on 1995/1997 land use/land cover (LULC) data, which was the most current data available at the time of our study; consideration should be given to updating both the value estimates and the maps when more recent LULC data become available.

For a number of reasons, the dollar amounts presented above are almost certainly conservative, i.e., they underestimate the true value of New Jersey's ecosystem services. These reasons include gaps in the valuation literature as well as a number of technical factors discussed at the end of the main text in this part of the report.

B. Hedonic Analysis

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 study, we analyzed the effect on actual residential housing prices of proximity to several environmental amenities, including beaches, protected open space (specifically, large, medium and small parks), water bodies, and unprotected forests and wetlands.

To ensure that the effects being attributed to proximity to environmental amenities are not in fact due to non-environmental factors, our analysis adjusted for many other factors related to residential housing prices, including lot size, number of rooms, property taxes, etc. Because this requires very detailed information on a large number of actual market transactions, and because such information is only readily available from commercial data vendors, resource limitations prevented us from conducting a hedonic analysis for the entire state. We therefore focused on seven local housing markets located in Middlesex, Monmouth, Mercer and Ocean Counties; in most respects those markets are demographically similar in the aggregate to the state as a whole.

We ran two types of hedonic analysis using this database. In the first, we defined proximity in terms of various mutually exclusive locational zones, e.g., a house is either within 300 feet of a beach or it is not; in this analysis, the exact distance is not taken into account. In the second type of analysis, we used the exact distance from the amenity, e.g., we distinguished between houses located 100 feet and 200 feet from a beach. Where the two analyses agree, we can have increased confidence in the results. We could not run all of the analyses in each of the seven real estate markets, either because a given market lacked the environmental amenity in question or because it had too few home sales involving that amenity to draw statistically valid conclusions.

The results we obtained in the two analyses demonstrate that homes that are closer to environmental amenities generally sell for more than homes further away, all else being equal. We first present the results based simply on whether a home is within a given distance of an environmental amenity or not:

1. Beach zones (7 markets analyzed). In four markets, sale prices for homes within 300 feet of a beach were from \$81,000 to \$194,000 higher than homes further away. For two markets, homes between 300 and 2,000 feet from a beach had prices that were from \$16,000 to \$44,000 higher than homes further away; however, in one market the selling price for a home so located was \$28,000 *lower*, presumably reflecting market-specific factors not controlled for.

2. Environmentally Sensitive (ES) zones (2 markets analyzed). Houses located in ES zones, as defined by the Office of State Planning, had selling prices that were between \$8,600 and \$34,500 higher than houses not located in such zones.

3. Water zones (1 market analyzed). Houses located within 100 feet of a water body sold for

$33,000 more than homes not so located.

These results show that whether or not a property is located within a given distance of an environmental amenity affects a home's value as measured by its sale price. As noted above, we also tested the impact of the exact distance to amenities, but those results were much less clear. The summary below gives results for two specific distances—100 feet and 5 miles—but results for other distances can also be generated.

1. Proximity to <u>beaches</u> was consistently positively valued. For example, in the two markets we were able to analyze in terms of exact distance to beaches, homes located 100 feet from a beach sold for between $13,000 and $21,000 more than homes located 5 miles away from the beach, with smaller increases in value for homes located at intermediate distances.

2. Proximity to <u>water features</u> was positively valued in two markets, with homes located 100 feet from a water feature selling for between $32,000 and $92,000 more than homes located 5 miles away from the feature. However, in a third market, homes located 100 feet away sold for over $63,000 <u>less</u> than homes 5 miles away, presumably reflecting local factors not captured in the analysis.

3. <u>Unprotected forests and wetlands</u> were consistent in having no strong effects on property values across markets.

4. The market value of proximity to <u>parks</u> varied depending on the size of the park:

   - Proximity to small parks (< 50 acres) was positively valued in four markets (prices between $17,000 and $178,000 higher at 100 feet from the park than at 5 miles) and negatively in one market (selling prices $86,000 *lower* at 100 feet than at 5 miles).

   - Proximity to medium parks (50-2,000 acres) was valued positively in two markets (price difference between $9,000 and $66,000) and negatively in four (price difference between -$19,000 and -$272,000).

   - Proximity to large parks (> 2,000 acres) was valued positively in three markets (price difference between $33,000 and $40,000) and negatively in another three (price difference between -$25,000 and -$176,000).

While we can say that proximity to small parks tends to have a consistently positively effect on housing prices, the mixed or negative results for proximity to medium and large parks are harder to explain other than as the results of confounding effects of unidentified negative factors associated with large open space areas in local housing markets. For example, in some markets medium or large parks might be located further from stores, transportation, or job opportunities. Identification of such confounding factors would require further analysis and resources.

A recognized inherent limitation of hedonic analysis is that the results cannot be readily translated into dollar values per acre and so are difficult to compare with the results of the value transfer analysis. The limited tests we were able to perform to address this problem suggest that the valuations obtained from the hedonic analysis translate into *larger* per acre dollar amounts than we obtained from the value transfer analysis, suggesting that the latter may be conservative, i.e., on the low side.

C. Spatial Modeling

Spatial modeling, as applied in this study uses a landscape simulation model to assess the relationships over time between specific spatial patterns of land use and the production of ecosystem services. We used a model that has been previously designed, calibrated, and thoroughly tested for a watershed in Maryland. While the *absolute* results for watersheds in New Jersey could be substantially different, the *relative* values for ecosystem services in various scenarios are likely to be consistent.

In this analysis we tracked two variables related to ecosystem services: (1) concentration of nutrients (in this case nitrogen), an important indicator of water quality; and (2) Net Primary Productivity (NPP), a proxy for total ecosystem services value. (NPP essentially measures the amount of plant growth and is therefore an indicator of the amount and health of existing vegetation; since animal food webs rely ultimately on vegetation, NPP also measures the growth rate for the resources on which animal life depends.) The model includes variables that can quantify how much these indicators may vary as land use, climate, and other factors change in spatial location and over time.

Our results show that different land use allocations and spatial patterns affect the ecosystem services generated. For the water quality index, this difference can be as large as 40%. Forests located close to a river's estuary zone contribute more to estuary water quality than forests located further away. Further, small river buffers have only a minor impact on water quality and need to be fairly large to be of use, whereas small, dispersed forest patches do more to enhance water quality than larger forest clusters. There is still much uncertainty in these estimates, and more detailed and comprehensive studies are required to take into account the whole set of ecosystem services and to account properly for the precise spatial variations in land cover and location, but these results show that spatial patterns of land use can affect ecosystem services significantly.

Conclusions

1. Ecosystems provide a wide variety of economically valuable services, including waste treatment, water supply, disturbance buffering, plant and animal habitat, and others. The services provided by New Jersey's ecosystems are worth, at a minimum, $11.6-19.4 billion/year. For the most part, these services are not currently accounted for in market transactions.

2. These annual benefits translate into a present value for New Jersey's natural capital of at least $387 billion to $648 billion, not including marketed ecosystem or abiotic goods or secondary economic impacts.

3. Wetlands (both freshwater and saltwater), estuaries/tidal bays, and forests are by far the most valuable ecosystems in New Jersey's portfolio, accounting for over 90% of the estimated total value of ecosystem services.

4. A large increase in property values is associated with proximity to beaches and open water. Proximity to smaller urban and suburban parks has positive effects in most markets, while the value of proximity to larger tracts of protected open space and environmentally sensitive areas depends on the local context.

5. Landscape modeling shows that the location of ecosystems relative to each other significantly affects their level of ecoservice production.

6. Significant gaps exist in the valuation literature, including gas and climate regulation provided by wetlands; disturbance prevention provided by freshwater wetlands; disturbance prevention, water supply, and water regulation provided by forests; and nutrient regulation, soil retention/formation, and biological control provided by a number of ecosystems.

7. While the assessment is far from complete and probably can never be considered final, the general patterns are clear and should receive careful consideration in managing New Jersey's ecosystems and other natural capital to preserve and enhance their long-term value to society.

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Overview of the Study

The New Jersey Context

Between 1986 and 1995, New Jersey converted almost 149,000 acres or almost 4.4% of its forests, farmland, and wetlands to other uses; this works out to 16,545 acres annually or about 0.5%.[1] Acting as individuals, through the private sector, and through their elected and appointed public officials, New Jerseyans are making decisions on a daily basis on the future of their remaining natural environment, and issues involving development and land use are at or near the top of the list of public issues of concern to New Jerseyans.

In making these decisions, New Jersey's residents and public officials are constantly choosing between competing uses of the "natural" environment.[2] Such choices usually (although not always) involve a choice between preserving land in its existing state or converting it to residential or commercial use, including built infrastructure such as roads and highways.

- Should a patch of forest be cleared to provide new land for roads, or should it be maintained in its current state to serve as a recreational resource? About 62,000 acres of forest were cleared for development (or cleared and left barren) between 1986 and 1995, net of developed or barren land that was converted to forest through tree planting programs.
- Should a particular wetland be drained and developed for commercial purposes or maintained "as is" to serve as a wildlife habitat and storm water buffer? Some 22,000 acres of wetland were developed or rendered barren between 1986 and 1995.
- Should a parcel of farmland be sold for housing development or preserved for farming? From 1986 to 1995, about 65,000 acres of farmland were developed or rendered barren. (Another 22,000 acres of farmland were allowed to revert to forest during that period.)

While making choices among these competing land use alternatives does not turn solely on economic considerations, it is obviously essential to have a broad understanding of both the benefits and the costs of development. The benefits usually attributed to development by its proponents are well-known, including provision of housing, economic development, job creation, improving transportation infrastructure, strengthening municipal finances, etc. Some of the costs of development are equally familiar, including increased demand for municipal services, public infrastructure, costs for school system expansion, traffic congestion and longer daily commutes, stress on water supplies, and so forth.

While the benefits of environmental preservation and the environmental costs of development are also familiar—land conversion and the loss of natural features that were previously part of a landscape—they are often not treated in economic terms in the same sense as, say, the cost of a new school or highway. Many of the social and ecological costs of development, including degradation of water quality, silting of rivers and streams, increasing levels of air pollution, and so on, are simply left out of the analysis of the trade-offs accompanying land use decisions. The environmental benefits of preservation—which in many cases are the converse of the costs of development—are often similarly ignored.

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

[1] The source for all land use and land cover data cited in this section is Hasse and Lathrop (2001). The 1986 and 1995 data used in that source are the most recent official data available on these subjects. Land use and land cover data for 2002 are expected to become available sometime in 2006.

[2] It can be argued that farmland is not "natural" in the same sense as unmanaged forests and wetlands. For purposes of this report, however, farmland is more akin to such landscapes than to urbanized areas.

gaining when we preserve a landscape in its undeveloped state or what we lose when we decide (deliberately or by default) 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 landscapes and the human values they represent as explicit and transparent as possible.

This need for information is not limited to environmental issues. For *any* efficient market transaction or public policy decision, both theory and common sense tell us that costs and benefits need to be made transparent to agents; if the market is not transparent; inefficiencies arise because people make uninformed choices leading to suboptimal or "irrational" decisions (Shiller, 2000). The identification and measurement of environmental features of value is thus essential for the efficient and rational allocation of environmental "resources" among the competing demands on natural and cultural landscapes (Daily, 1997; Costanza et al., 1997; Wilson & Carpenter, 1999).

This project aims to present a comprehensive assessment of the economic benefits provided by New Jersey's natural environment. Our goal was to use the best available conceptual frameworks, data sources, and analytic techniques to generate value estimates that can be integrated into land use planning and environmental decision-making throughout New Jersey. By estimating the economic value of environmental features not traded in the marketplace, social costs or benefits that otherwise would remain hidden or unappreciated are revealed, so that when tradeoffs between alternative land uses in New Jersey are evaluated, information is available to help decision makers avoid systematic biases and inefficiencies.

Definitions and Ethical Concerns

Before discussing the value of benefits[3] provided by the natural environment, we need to clarify some underlying concepts and terms. The following definitions are based on Farber et al. (2002).

"Value systems" refer to the 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 result from 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 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. If individuals have good knowledge of an object or action's connection to their well-being, one can use their "willingness-to-

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[3] As used in this and similar contexts throughout this report, "environmental benefits" means the benefits that the natural environment provides to human beings, either directly or indirectly (e.g., retention of soil by forests), rather than the benefits "to the environment" from controlling pollution, (e.g. reduced particulate emissions from combustion of diesel fuel.)

pay" for the object or action as a measure of its value to them. This willingness to pay can be either revealed through their actions (i.e. housing market choices as in the hedonic analysis discussed later) or stated as a response to surveys of various kinds (i.e. contingent value surveys of the type used in some of the value transfer studies discussed later).

"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 or end in itself. This is sometimes referred to as being "biocentric" rather than "anthropocentric." "Values", (as defined above) are based on the contribution that something makes to achieving goals (directly or indirectly), i.e., they represent *instrumental values*. 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. One should 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. Since an intrinsic value is a goal or end, one cannot measure or quantify the "intrinsic value" of something.

In modern economics the term value is usually taken to mean "*exchange value"*, defined as the maximum amount that an individual would be willing to pay to obtain a benefit or the minimum that the person would be willing to accept to forego the benefit. The data accepted as providing evidence of the amount of value in this sense are often restricted to stated or revealed preferences, but one can (and must, if one hopes to be comprehensive and accurate) encompass valuations from multiple perspectives, using multiple methods (including both subjective and objective), against multiple goals (Costanza, 2000).

Some environmentalists object on principle to assigning economic values to nature. The objection seems to be that it is somehow "unethical" or "vulgar" or self-defeating to attempt to quantify environmental benefits in dollar terms. This type of objection is difficult to address except by saying we see no logical conflict between identifying economic reasons for preserving natural systems and stating ethical reasons; in principle, these are mutually supportive rather than either/or justifications.

The objection may be based partly on the false presumption that quantifying dollar values for natural "assets" automatically implies that they can or should be traded in private markets. However, natural assets are, for the most part, public goods. They are often "non-rival" (one person's use does not preclude other's use) and "non-exclusive" (it is difficult or impossible to exclude people from benefiting from the services). These characteristics are the economist's classic criteria for "public" goods, and most economists would agree that using unfettered private markets to manage these assets will not maximize social welfare.

In common with conventional "manufactured" public goods such as roads, bridges, and other publicly-owned infrastructure, a significant government involvement in the production and management of environmental benefits is therefore necessary. However, just because we decide that we cannot or should not sell a public asset such as the Brooklyn Bridge does not mean we should not quantify its value. Effectively managing and maintaining the bridge requires knowledge of its social costs and benefits, and the same reasoning applies to managing our endowment of natural assets.

The objection may also be based on the idea that "there are some things you can't [and by implication shouldn't] put a price on". While it is certainly true that there are some things we probably never would (or should) sell for money, this is not the same as saying that it is unethical to assign a value (expressed in dollar terms) to some aspect of nature that we value, e.g., preservation of habitat for the bald eagle or another rare species. The alternative to doing so—leaving a blank space in that part of the analysis—is in effect to accept an implicit value of zero in discussing the costs and benefits of preserving that habitat. Saying that the value is infinite or "beyond money" leads to much the same result—the "space" is left blank, albeit with an explanation that the good or service in question cannot be valued.

In our world, resources are always limited, and the resources devoted to habitat preservation can always find other worthy uses. When one alternative is chosen over another, e.g. development vs. preservation of a particular habitat, the choice indicates which alternative is deemed to be worth the most, i.e., which is more valuable. Therefore, "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). Of course, it may be very difficult (given our present knowledge) to assign a defensible value to some aspects of the environment. However, the record in this field (cf. Appendix A) has been one of development and refinement of valuation methods to address such challenges, and the only way to know whether something can be usefully valued is to make the attempt.[4]

Environmental Sources of Economic Benefits

In earlier eras, economic benefits associated with the natural environment were 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 human beings. In general, the inanimate resources are non-renewable, i.e., they are potentially exhaustible, although exploration may uncover new sources and technological development may create substitutes. Animate resources, on the other hand, are potentially renewable if they are not harvested too rapidly and if other factors (e.g., climate, absence of disease, etc.) are favourable to their renewal.

A different way of looking at environmental benefits has been gaining favor over the last several decades among scientists and economists. 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 inanimate or "abiotic" resources are not ignored, the emphasis is on the benefits provided by the living environment, usually viewed in terms of whole ecosystems. Ecosystems are 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, Costanza and Troy, 2004). The recently released Millennium Ecosystem Assessment represents the work of over 1300 scientists worldwide over four years focused on the concept of ecosystem services and their contribution to human well-being (http://www.millenniumassessment.org/en/index.aspx)

The New Jersey landscape is composed of a diverse mixture of forests, grasslands, wetlands, rivers, estuaries and beaches that provide many different valuable goods and services to human beings. Ecosystem *goods* represent the material products that are obtained from nature for human use (De Groot et al., 2002), such as timber from forests, fish from lakes and rivers, food from soil, etc. An ecosystem *service*, in contrast, consists of "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life" (Daily, 1997).

[4] Where the benefits of an action are especially difficult to quantify in monetary terms, benefit-cost analysis may have to give way to cost-effectiveness analysis, where the end—e.g. habitat preservation—is taken as a given and the analyst and policymaker look for the least-cost means of achieving that end. In general the present report does not address the costs of environmental preservation, and it therefore represents yet another approach, namely valuation of the natural assets at stake in land use decisions.

The ecosystem services that we evaluate in this project are listed below[5]:

1. ***Climate and atmospheric gas regulation****:* life on earth exists within a narrow band of chemical balance in the atmosphere and oceans, and alterations in that balance can have positive or negative impacts on natural and economic processes. Biotic and abiotic processes and components of natural and semi-natural ecosystems influence this chemical balance in many ways including the CO_2/O_2 balance, maintenance of the ozone-layer (O_3), and regulation of SO_X levels.
2. ***Disturbance prevention****:* many natural and semi-natural landscapes provide a 'buffering' function that protects humans from destructive perturbations. For example, wetlands and floodplains can help mitigate the effects of floods by trapping and containing stormwater. Coastal island vegetation can also reduce the damage of wave action and storm surges. The estimated cost of floods in the U.S. in terms of insurance claims and aid exceed $4 billion per year.
3. ***Freshwater regulation and supply****:* the availability of fresh and clean water is essential to life, and is one of humanity's most valuable natural assets. When water supplies fail, water must be imported from elsewhere at great expense, must be more extensively treated (as in the case of low stream flows or well levels), or must be produced using more expensive means (such as desalinization). Forests and their underlying soil, and wetlands, play an important role in ensuring that rainwater is stored and released gradually, rather than allowed to immediately flow downstream as runoff.
4. ***Waste assimilation****:* both forests and wetlands provide a natural buffer between human activities and water supplies, filtering out pathogens such as *Giardia* or *Escherichia*, nutrients such as nitrogen and phosphorous, and metals and sediments. This service benefits both humans by providing cleaner drinking water and plants and animals by reducing harmful algae blooms, increasing dissolved oxygen and reducing excessive sediment in water. Trees also improve air quality by filtering out particulates and toxic compounds from air, making it more breathable and healthy.
5. ***Nutrient regulation***: the proper functioning of any natural or semi-natural ecosystem is dependent on the ability of plants and animals to utilize nutrients such as nitrogen, potassium and sulfur. For example, soil and water, with the assistance of certain bacteria algae (Cyanobacteria), take nitrogen in the atmosphere and "fix" it so that it can be readily absorbed by the roots of plants. When plants die or are consumed by animals, nitrogen is "recycled" into the atmosphere. Farmers apply tons of commercial fertilizers to croplands each year, in part because this natural cycle has been disrupted by intense and overly-extractive cultivation.
6. ***Habitat refugium***: contiguous 'patches' of landscape with sufficient area to hold naturally functioning ecosystems support a diversity of plant and animal life. As patch size decreases, and as patches of habitat become more isolated from each other, population sizes can decrease below the thresholds needed to maintain genetic variation, withstand stochastic events (such as storms or droughts) and population oscillations, and meet "social requirements" like breeding and migration. Large contiguous habitat blocks, such as intact forests or wetlands, thus function as critical population sources for plant and animal species that humans value for both aesthetic value and functional reasons.

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[5] Alternative lists of ecosystem goods and services have been proposed (see for example, Costanza et al., 1997 and De Groot et al., 2002); but we selected this list for its specific applicability to landscape analysis using available land cover and land use data.

7. ***Soil retention and formation****:* soils provide many of the services mentioned above, including water storage and filtering, waste assimilation, and a medium for plant growth. Natural systems both create and enrich soil through weathering and decomposition and retain soil by preventing its being washed away during rainstorms.

8. ***Recreation****:* intact natural ecosystems that attract people who fish, hunt, hike, canoe or kayak, bring direct economic benefits to the areas surrounding those natural areas. People's willingness to pay for local meals and lodging and to spend time and money on travel to these sites, are economic indicators of the value they place on natural areas.

9. ***Aesthetic and amenity***: Real estate values, and therefore local tax revenues, often increase for houses located near protected open space. The difference in real estate value reflects people's willingness to pay for the aesthetic and recreational value of protected open space. People are also often willing to pay to maintain or preserve the integrity of a natural site to protect the perceived beauty and quality of that site.

10. ***Pollination****:* More than 218,000 of the world's 250,000 flowering plants, including 80% of the world's species of food plants, rely on pollinators for reproduction. Over 100,000 invertebrate species — such as bees, moths, butterflies, beetles, and flies — serve as pollinators worldwide. At least 1,035 species of vertebrates, including birds, mammals, and reptiles, also pollinate many plant species. The US Fish and Wildlife Service lists over 50 pollinators as threatened or endangered, and wild honeybee populations have dropped 25 percent since 1990. Pollination is essential for many agricultural crops, and substitutes for local pollinators are increasingly expensive.

As the above listing indicates, ecosystem goods and services affect humanity at multiple scales, from climate regulation and carbon sequestration at the global scale, to flood protection, soil formation, and nutrient cycling at the local and regional scales (De Groot et al., 2002). They also span a range of degrees of connection to human welfare, with services like climate regulation being less directly or immediately connected, and recreational opportunities being more directly connected.

The concept of ecosystem services is useful for landscape management, sustainable business practice and decision making for three fundamental reasons. *First*, it helps us synthesize essential ecological and economic concepts, allowing researchers and managers to link human and ecological systems in a viable and relevant manner. *Second*, it draws upon the latest available ecosystem science. *Third*, public officials, business leaders and citizens can use the concept to evaluate economic and other tradeoffs between landscape development and conservation alternatives.

Driven by a growing recognition of their importance for human life and well-being, ecologists, social scientists, and environmental managers have become increasingly interested in assessing the economic values associated with both ecosystem goods and services (Bingham et al, 1995; Costanza et al., 1997; Farber et al., 2002) and increasingly skilled in developing and applying appropriate analytic techniques for performing those assessments.

Organization of This Report

Our approach to valuing New Jersey's ecosystem services includes four main components as follows:

1. A framework for classifying environmental benefits and the types of landscape that generate them;

2. A "value transfer" methodology for valuing ecosystem services that emphasizes that no single study alone can capture the total value of a complex ecological system;

3. A spatial context for landscape valuation using land cover data and Geographic Information Systems (GIS); and

4. An assessment of the effects of spatial pattern and proximity effects on ecosystem services and their value.

Our results include the following:

1. Tables synthesizing the results of more than 150 primary studies on the value of each ecosystem type and ecosystem service flow included in our study;
2. Tables compiling the value of ecosystem service flows for the entire state;
3. Maps of the current value of ecosystem service flows in New Jersey based on these estimates;
4. The results of a primary study of ecosystem amenity values we performed using New Jersey data and hedonic analysis techniques;
5. An analysis of the effects on ecosystem service values of differences in spatial patterns of land use; and
6. The results of converting annual flows of ecosystem service values to estimates of the value of New Jersey's stock of natural capital.

Methods and Results

Measuring Values for Ecosystem Services

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: *marketed* and *non-marketed.*

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, 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 are identified in Table 1.

As the descriptions in Table 1 suggest, each valuation methodology has its own strengths and limitations, often limiting its use to a select range of ecosystem goods and services within a given landscape. For example, the value generated by a naturally functioning ecological system in the treatment of wastewater can be estimated using the Replacement Cost (RC) method, which is based on the price of the cheapest alternative way of obtaining that service, e.g. the cost of chemical or mechanical alternatives. A related method, Avoided Cost (AC), can be used to estimate value based on the cost of damages due to lost services. Travel Cost (TC) and Contingent Valuation (CV) surveys are useful for estimating recreation values, while Hedonic Pricing (HP) is used for estimating property values associated with aesthetic qualities of natural ecosystems. In this project, we synthesized studies which employed the full suite of ecosystem valuation techniques. We also performed an original hedonic analysis of the relationship between property sales prices and ecological amenities.

Table 1: Non-Market Economic Valuation Techniques

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 (RC): 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, including the imputed value of their time.

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 a production function (i.e., Cobb-Douglas) to estimate the change in the value of outputs in response to a change in material inputs.

Contingent Valuation (CV): service demand may be elicited by posing hypothetical scenarios that involve some valuation of alternatives; e.g., people generally state that they 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.*

Value Transfer Approach

In this report, we use value transfer to generate baseline estimates of ecosystem service values in the state of New Jersey (Desvouges et al., 1998). Value transfer involves the adaptation of existing valuation information or data to new policy contexts[6]. In this analysis, the transfer method involves obtaining an economic estimate for the value of non-market services through the analysis of a single study, or group of studies, that have been previously carried out to value similar services. The transfer itself refers to the application of values and other information from the original 'study site' to a new 'policy site' (Desvouges et al., 1998; Loomis, 1992; Smith, 1992).

With the increasing sophistication and number of empirical economic valuation studies in the peer-reviewed literature, value transfer has become a practical way to inform decisions when primary data collection is not feasible due to budget and time constraints, or when expected payoffs are small (Kreuter et al., 2001; Moran, 1999). As such, the transfer method is a very important tool for policy makers since it can be used to reliably estimate the economic values associated with a particular landscape, based on existing research, for considerably less time and expense than a new primary study.

The value transfer method is increasingly being used to inform landscape management decisions by public agencies (Downing & Ozuna, 1996; Eade & Moran, 1996; Kirchoff et al., 1997; Smith, 1992). Thus, it is clear that despite acknowledged limitations such as the context sensitivity of value estimates, existing studies can and do provide a credible basis for policy decisions involving sites other than the study site for which the values were originally estimated. This is particularly true when current net present valuations are either negligible or (implicitly) zero because they have simply been ignored. The critical underlying assumption of the transfer method is that the economic value of ecosystem goods or services at the study site can be inferred with sufficient accuracy from the analysis of existing valuation studies at other sites. Clearly, as the richness, extent and detail of information increases within the source literature, the accuracy of the value transfer technique will likewise improve.

While we accept the fundamental premise that primary valuation research will always be a 'first-best' strategy for gathering information about the value of ecosystem goods and services (Downing and Ozuna, 1996; Kirchhoff, 1997; Smith, 1992), we also recognize that value transfer has become an increasingly practical way to inform policy decisions when primary data collection is not feasible due to budget and time constraints, or when expected payoffs are small (Environmental Protection Agency, 2000; National Research Council, 2004). When primary valuation research is not possible or plausible, then value transfer, as a 'second-best' strategy, is important to consider as a source of meaningful baselines for the evaluation of management and policy impacts on ecosystem goods and services. The real-world alternative is to treat the economic values of ecosystem services as zero; a status quo solution that, based on the weight of the empirical evidence, will often be much more error prone than value transfer itself.

Summary of the Value Transfer Approach

As Figure 1 below shows, the raw data for the value transfer exercise in this report comes from previously conducted empirical studies that measured the economic value of ecosystem services. These studies were reviewed by the research team and the results analyzed for value transfer to the State of New Jersey. By entering the original results into a relational database format, each dollar value estimate can be identified with unique searchable criteria (i.e., type of study, author, location, etc.), thus allowing the team to associate specific dollar estimates with specific conditions on-the-ground. For example, all forest-related value estimates in this report come from economic studies that were originally conducted in

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[6] Following Desvouges et al. (1998), we adopt the term 'value transfer' instead of the more commonly used term 'benefit transfer' to reflect the fact that the transfer method is not restricted to economic benefits, but can also be extended to include the analysis of potential economic costs, as well as value functions themselves.

temperate forests similar to those in New Jersey. To achieve this, once analyzed, the valuation data were integrated with land cover data for New Jersey. Tables and maps were then generated from this fusion of economic and geographic information.

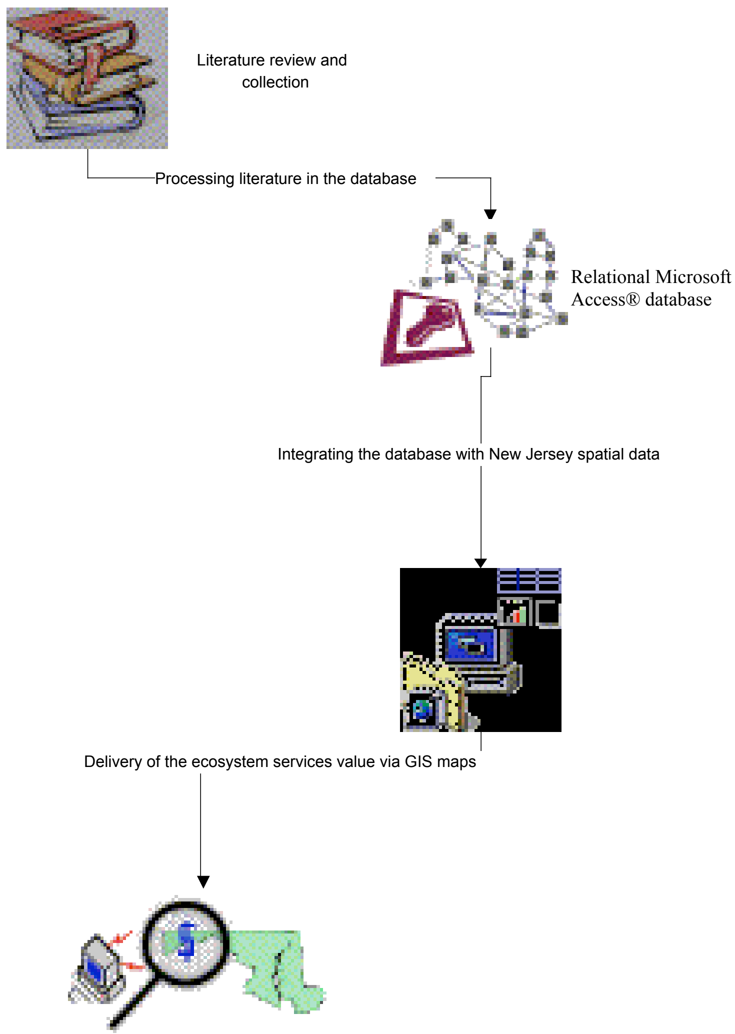

Figure 1: Stages of Spatial Value Transfer

The research team developed a set of decision rules for selecting empirical studies from the literature that allowed us to estimate the economic value of ecosystem services in the state of New Jersey. Using scientific data search engines such as ISI Web of Science® and by cross-checking the largest value transfer database online (i.e., EVRI ™) the research team reviewed the best available economic literature and selected valuation studies which were:

- Focused on temperate regions in North America
- Focused primarily on non-consumptive use

The quality of original studies used in the value transfer exercise always determines the overall quality and scope of the final value estimate (Brouwer, 2000). In our review of the literature, we were able to identify three general categories of valuation research, each with its own strengths and weaknesses (Table 2). Type A studies are peer-reviewed empirical analyses that use conventional environmental economic techniques (e.g., Travel Cost, Hedonic Pricing and Contingent Valuation) to elicit individual consumer preferences for environmental services. Type B studies are commonly referred to as the '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 that 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.

Table 2: Value-Transfer Data Source Typology

Type A	Type B	Type C
• Peer-Reviewed Journal Article or Book Chapter	• Non Peer-Reviewed (PhD Thesis, Raw Data, Technical Report etc.)	• Secondary (meta) Analysis of Peer reviewed and Non Peer Reviewed studies
• Uses Conventional Environmental Economic Valuation Methods	• Uses Conventional Environmental Economic Valuation Methods	• Uses Both Conventional and Non-Conventional Valuation methods
• Restricted to conventional, Preference-based Values	• Restricted to conventional, Preference-based Values	• Includes conventional Preference-based, non-conventional preference-based, and Non-Preference-based Values

The research team used two alternative approaches to capture possible variation in results across the different literature types: (1) we first limited our value transfer analysis to peer-reviewed studies that use conventional environmental economic methods (hereafter Type A studies) and (2) we then added a few additional Type B studies and Type C meta-analyses of ecosystem service values that were readily accessible (hereafter Type A-C). The results presented below are separated into Type A and Type A-C categories to generate a more complete picture of the complete range of ecosystem service values associated with the New Jersey landscape. For specific information on all the studies included in this report please see technical appendices B and C.

Land Cover Typology

Since ecosystem services are analyzed at the landscape scale for this project, a key challenge for the research team is to link the ecosystem service estimates to available land cover/land use data in New Jersey so that we can map ecosystem services (Wilson et. al. 2005). Thanks to the increased ease of using Geographic Information Systems (GIS) and the availability of land cover data sets derived from satellite images, ecological and geographic entities can more easily be associated with ecosystem services and the values they provide to people.

In simplified terms, the technique used to generate average ecosystem service value for a given geographic area involves combining one land cover layer with another layer representing the geography to which ecosystem services are aggregated – e.g., a watershed. While the aggregation units themselves are likely to be in vector format, because vector boundaries are most precise, the land cover layer may be either raster or vector.[7] Spatial disaggregation by watershed increases the contextual specificity of ecosystem value transfer by allowing us to visualize the exact location of ecologically important landscape elements and overlay them with other relevant themes for analysis—biogeophysical or socioeconomic.

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.[8] The new typology condenses a number of DEP classes that have similar (or no) ecosystem service value and creates several new classes to reflect important difference 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.

Table 3: 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

[7] The vector data model represents spatial entities with points, lines and polygons. The raster model uses a Cartesian grid to represent a landscape.

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

Figure 2: Land cover map of New Jersey

Most categories in this typology represent aggregations of pre-existing categories drawn from the NJDEP LULC map. For instance, the "beach" category in the new typology includes both the "beach" and the "vegetated coastal dunes" categories. However, several categories were developed using ancillary data sources in combination with the DEP land use/land cover map.

1. The first of these was the pasture/agricultural grassland category. While the NJDEP LULC map has a single category for both row crop agriculture and pastureland or hayfields, the valuation database contains studies differentiating between these two categories. To make this distinction geographically, we conducted overlay analysis between the LULC layer and the DEP's grasslands layer (which was based on a combination of the LULC data and data on sightings of imperiled or endangered grassland species). All map polygons designated as agricultural in the LULC that had their geometric centers within a grasslands polygon were designated as pasture/grassland.

2. The second category requiring ancillary data was urban greenspace. This layer was created by overlaying the urban centers boundary layer from the New Jersey Office of Smart Growth with the LULC map. All forest, wetland, and grassland polygons whose center fell within an urban center boundary were selected and recoded as urban green space.

3. The third category requiring ancillary data was riparian zones. This layer was created by buffering the DEP's layer of third order or higher rivers. One hundred foot-buffers were created around rivers of fourth and fifth order, and fifty foot buffers were created around third order watercourses. A geometric union overlay was then conducted between the LULC and buffer layers. All resulting polygons falling within the buffer were classified as riparian unless they were coded as wetlands, which was given precedence due to its higher ecoservice value.

4. The last category requiring ancillary data was the coastal shelf, which was created using the three-mile territorial waters buffer polygons present in the DEP's HUC-14 watershed management area layer.

Value Transfer Analysis - Results

Using the list of land cover classes described above, the research team conducted queries of the best available economic valuation data to generate baseline ecosystem service values estimates for the entire study area in New Jersey.

The research team obtained data from a set of 94 viable Type A studies and 100 viable Type A-C studies; as some publications provided multiple values, we were able to obtain 163 and 210 individual ESV estimates respectively (see below Tables 8 and 9). All results were standardized to average 2004 U.S. dollar equivalents per acre/per year to provide a consistent basis for comparison below[9]. The aggregated baseline ESV results for all land cover types represented within the study area are presented below in Tables 4 and 5.

Each table presents standardized average ESV data for ecosystem services associated with each of the unique land cover types generated in this analysis. For purposes of clarity and following convention in the literature (e.g., Costanza et. al. 1997; Eade and Moran 1999; Wilson et. al. 2005) all results presented in this report 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 tables 7 and 8 and detailed information for the literature sources used to calculate estimates for each ecosystem service-land cover pair is provided in technical Appendix B.

Moreover, for purposes of transparency, in addition to presenting a single point estimate for each land cover/ecosystem service pair, the minimum, maximum and alternative median dollar values are published for further review in Appendix C. As these technical tables reveal, statistical 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 statistical mean for beach ESV is approximately forty two thousand dollars per acre per year, while the statistical median is thirty eight thousand for both Type A and Type A-C studies, 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 statistical means were replaced with statistical medians[10].

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[9] All economic valuation data in this report are have been standardized to represent total net present values, not discounted. This allows for the results to be incorporated into forward looking scenarios that might weight future costs and benefits differently when summing over time (Heal, 2004).

[10] While it may also be tempting to narrow statistical ranges by discarding high and low 'outliers' from the literature, the data used in this section of the report were all 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.

Value Transfer Tables

The valuation results in Table 4 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 reveal that, there is a fairly robust spread of ESV's 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[11]. 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 4 also reveal considerable variability between averages ESV's delivered by different ecosystem service *types* in New Jersey. Once each average ESV is multiplied by the area of land cover type that provides it and summed across possible combinations, both water regulation and aesthetic/recreational services clearly stand out as the largest ecosystem service contributors to 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.

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

Value Transfer Tables

Table 4: Summary of average value of annual ecosystem services (Type A)

Ecosystem Services (2004 US$ $acre^{-1}\ yr^{-1}$)

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

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.

Table 5: Summary of average value of annual ecosystem services (Type A-C)

Ecosystem Services (2004 US$ $acre^{-1}$ yr^{-1})

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				521		723			20			35	$1,299
Beach	7,837		27,276									14,847	*24*	$42,147
Estuary	455,700		286		49		10,658			39	314	292	15	$11,653
Saltwater Wetland	190,520		310					5,413			201	26	180	$6,131
Terrestrial	**4,590,281**													
Forest	1,465,668	54			163	5		44	162	2	923	122	1	$1,476
Grass/Rangelands	583,009	3		2		3		44	13	12		1		$77
Cropland	90,455								8	12	831	15		$866
Freshwater Wetlands	814,479	134	3,657	2,986	1,544			838			113	1,406	890	$11,568
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	247,419,233	3,383,364,105	2,434,015,054	1,738,649,004	9,249,760	5,073,680,354	1,803,819,315	245,781,449	34,692,849	1,701,061,233	1,993,241,115	777,821,072	19,442,794,544

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.

The results in Table 5 were generated from the 94 Type A studies in Table 4, augmented by 6 additional Type B and Type C synthesis studies documented in the technical appendices. Even with the addition of this ESV data, beaches continue to provide the highest annual value per year ($42,147) while grassland/rangeland ($77) provides the lowest annual value. However, some interesting differences in results between tables 5 and 6 can be identified. For example, here the land cover category estuaries ($11,653) moves forward in overall rank from the eighth most valuable land cover class to the second most valuable. This shift appears to be driven primarily by the nutrient regulation service ($10,658) documented by the Costanza et al. (1997) synthesis study in Appendix C. Similarly, while Freshwater Wetlands ($11,568) and Saltwater Wetlands ($6,131) retain their overall high ranking in terms of ESV delivery, the addition of the synthesis study results appear to increase the magnitude of their annual ESV's substantially.

The column totals at the bottom of Table 5 again reveal considerable variability between average ESV flows delivered by different ecosystem service *types* in New Jersey; but it is also clear that the addition of synthesis studies and non-peer reviewed analysis have filled in some of the gaps documented above in Table 4. Once each average ESV is multiplied by the area of land cover type that provides it and summed across possible combinations, it appears that both nutrient cycling and disturbance regulation services stand out as the largest ecosystem contributors to annual ESV's in New Jersey, cumulatively representing over 8 billion in annual value. As mentioned above, the largest shift appears to be driven by the nutrient regulation service documented by Costanza et. al. (1997). At the other end of the spectrum, pollination and cultural services appear to contribute the least value to New Jersey.

Spatially Explicit Value Transfer

Once specific land cover types were identified, ecosystem service flow values for land cover types in New Jersey were determined by multiplying areas of each cover type, in acres, by the estimated annualized dollar value *per acre* for that cover type. The economic values used to estimate the values associated with each ecosystem good or service were drawn from the value transfer exercise as described above.

The total ESV of a given land use/land cover type for a given unit of analysis (i.e., watershed) were thus be determined by adding up the individual, ecosystem service values associated with each land use/land cover type. The following formula is used:

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

Where:

$A(LU_i)$ = Area of Land cover (i)

$V(ESV_i)$ = Annual value of Ecosystem Services (k) for each Land Use (i).

Resulting values were estimated for each land cover type in New Jersey using the value transfer methods described above. Total ESV flow estimates for each land cover category were estimated by taking the product of total average per acre service value and the area of each land cover type in the state. These results are summarized below in Table 6. 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 output of the operation included the area and the land cover type for each subwatershed. Maps were then created using graduated color classification to show both per acre and total ESV estimates for all New Jersey subwatersheds.

Table 6: Total Acreage and Mean Flow of Ecosystem Services in New Jersey

Name	Acreage	ESV Flows using A studies	ESV Flows using A-C studies
Coastal and Marine			
Coastal Shelf	299,835	$185,843,730	$389,455,682
Beach	7,837	$330,322,259	$330,322,259
Estuary and Tidal Bay	455,700	$325,989,335	$5,310,478,189
Saltwater Wetland	190,520	$1,243,545,862	$1,168,014,271
Terrestrial			
Forest	1,465,668	$1,880,935,494	$2,163,384,341
Pasture/grassland	583,009	$6,751,242	$44,623,493
Cropland	90,455	$2,103,089	$78,302,761
Freshwater Wetland	814,479	$7,081,746,098	$9,421,727,249
Open Fresh Water	86,232	$65,993,537	$65,993,537
Riparian Buffer	15,146	$51,230,004	$51,230,004
Urban Greenspace	169,550	$419,227,482	$419,227,482
Urban or Barren	1,365,742	$0	$0
TOTAL	5,544,173	**$11,593,688,132**	**$19,442,759,268**

Here, the data clearly show that substantial economic values are being delivered to New Jersey citizens every year by functioning ecological systems on the landscape. The estimated range is from a lower bound of approximately ***$11 billion per year*** to an upper bound of over ***$19 billion per year*** depending on the source literature used. Consistent with the value transfer data reported above in Table 4 and Table 5, it appears that ecosystem services associated with both freshwater and saltwater wetland types as well as forest and estuaries tend to provide the largest cumulative economic value.

As the following maps of New Jersey show (Figures 3-6), 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. This general pattern of spatial heterogeneity holds true for both Type A value-transfer results and Type A-C value transfer results suggesting that underlying differences are due to underlying landscape patterns on the ground. 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. This pattern is true for both the Type A study maps and Type A-C study maps. Similarly, when watersheds with considerable estuarine and tidal features are considered, the difference between Type A and Type A-C study stands out in sharp contrast with such watersheds consistently ranking highest in value in Figure 5 and Figure 6.

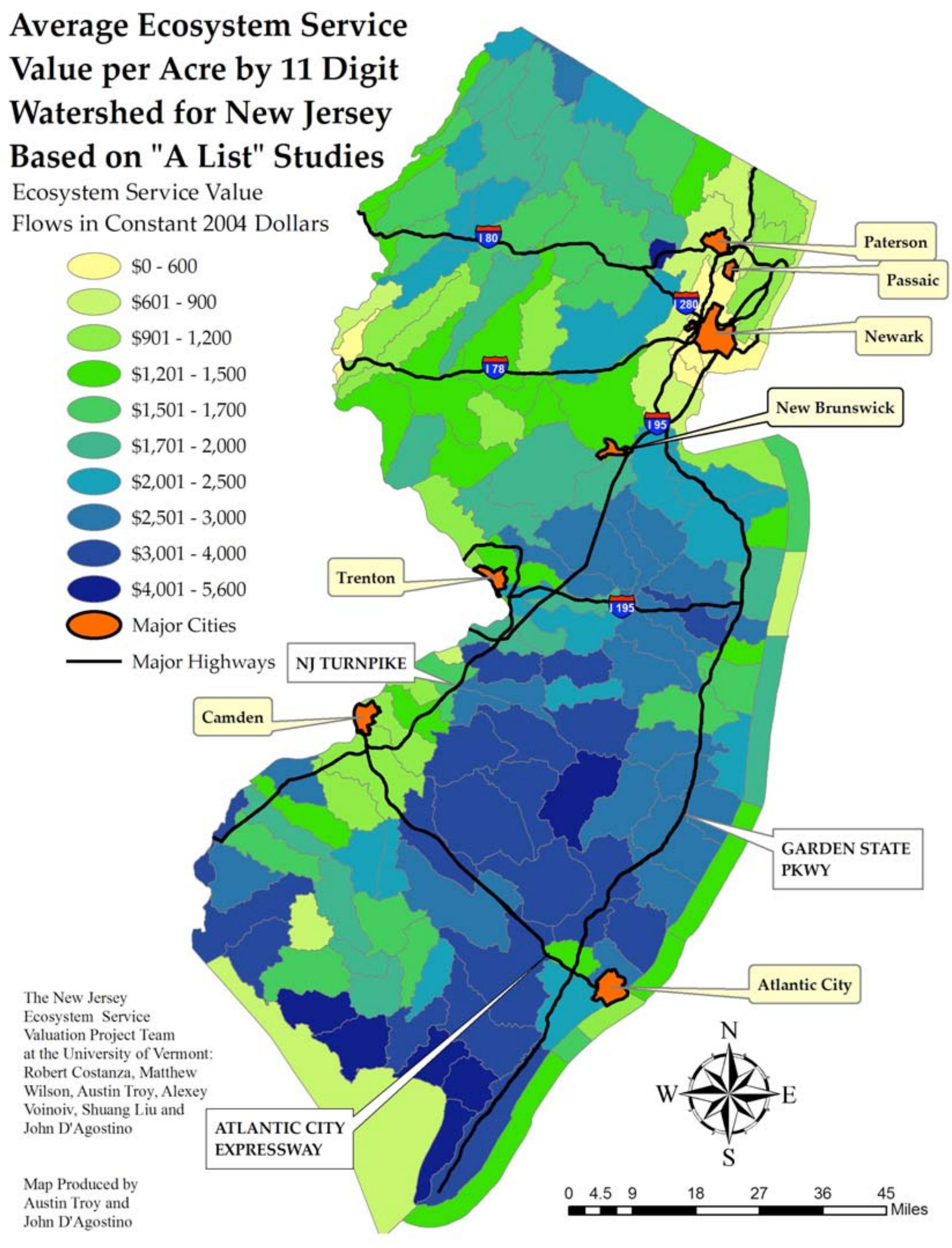


Figure 3: Average Ecosystem Service Value per acre by watershed for New Jersey based on Type A studies

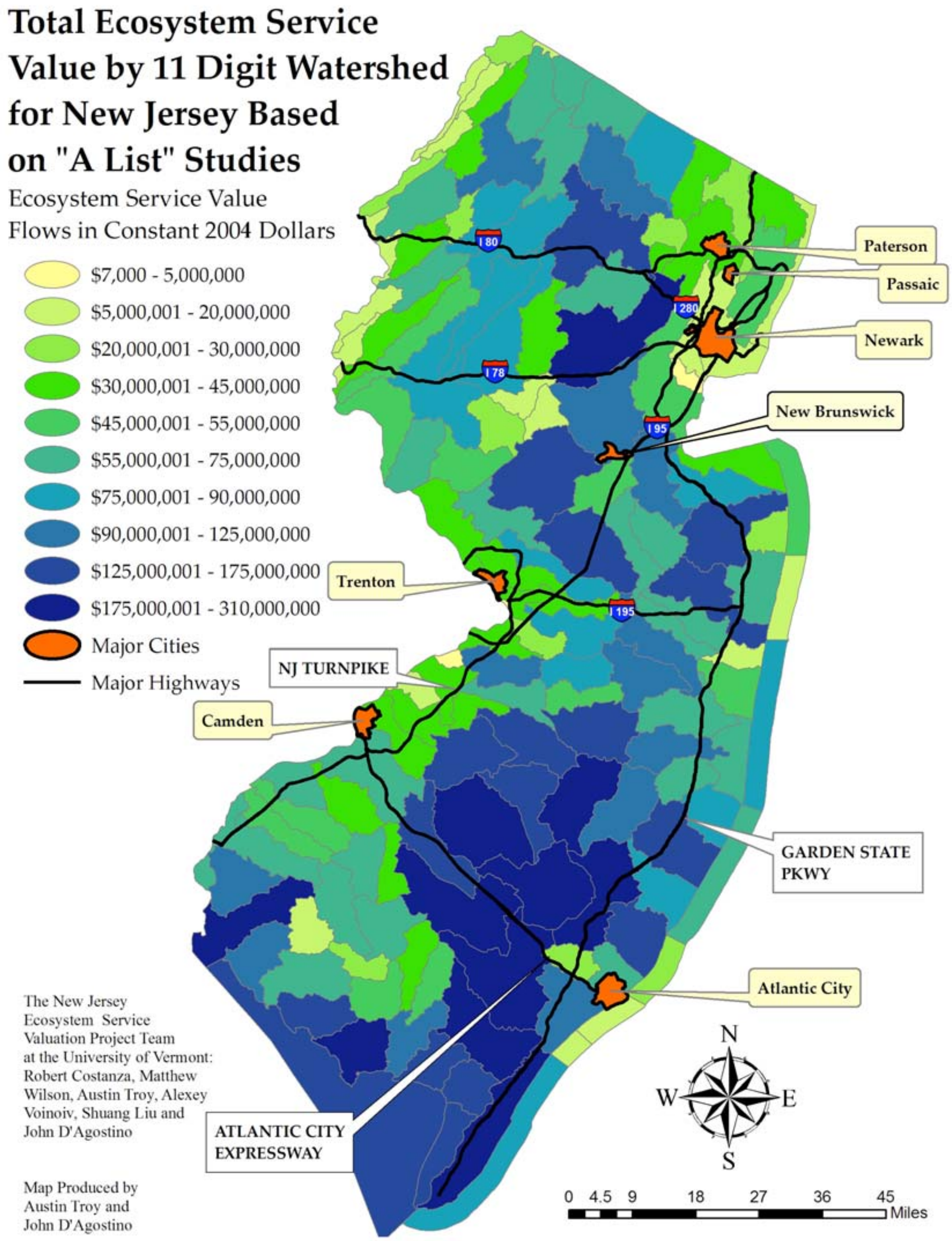


Figure 4: Total Ecosystem Service Value by watershed for New Jersey based on Type A studies

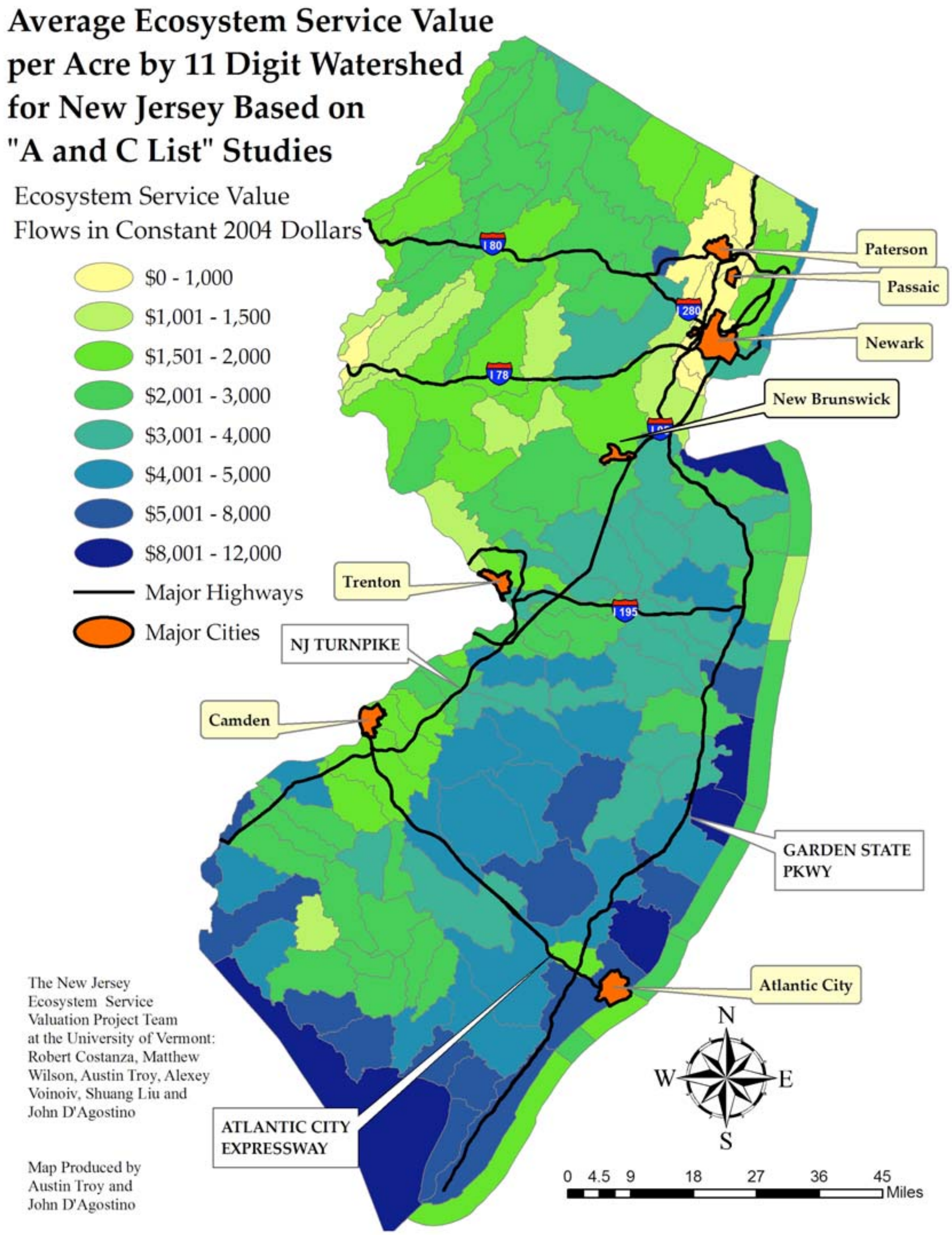


Figure 5: Average Ecosystem Service Value per acre by watershed for New Jersey based on Type A-C studies

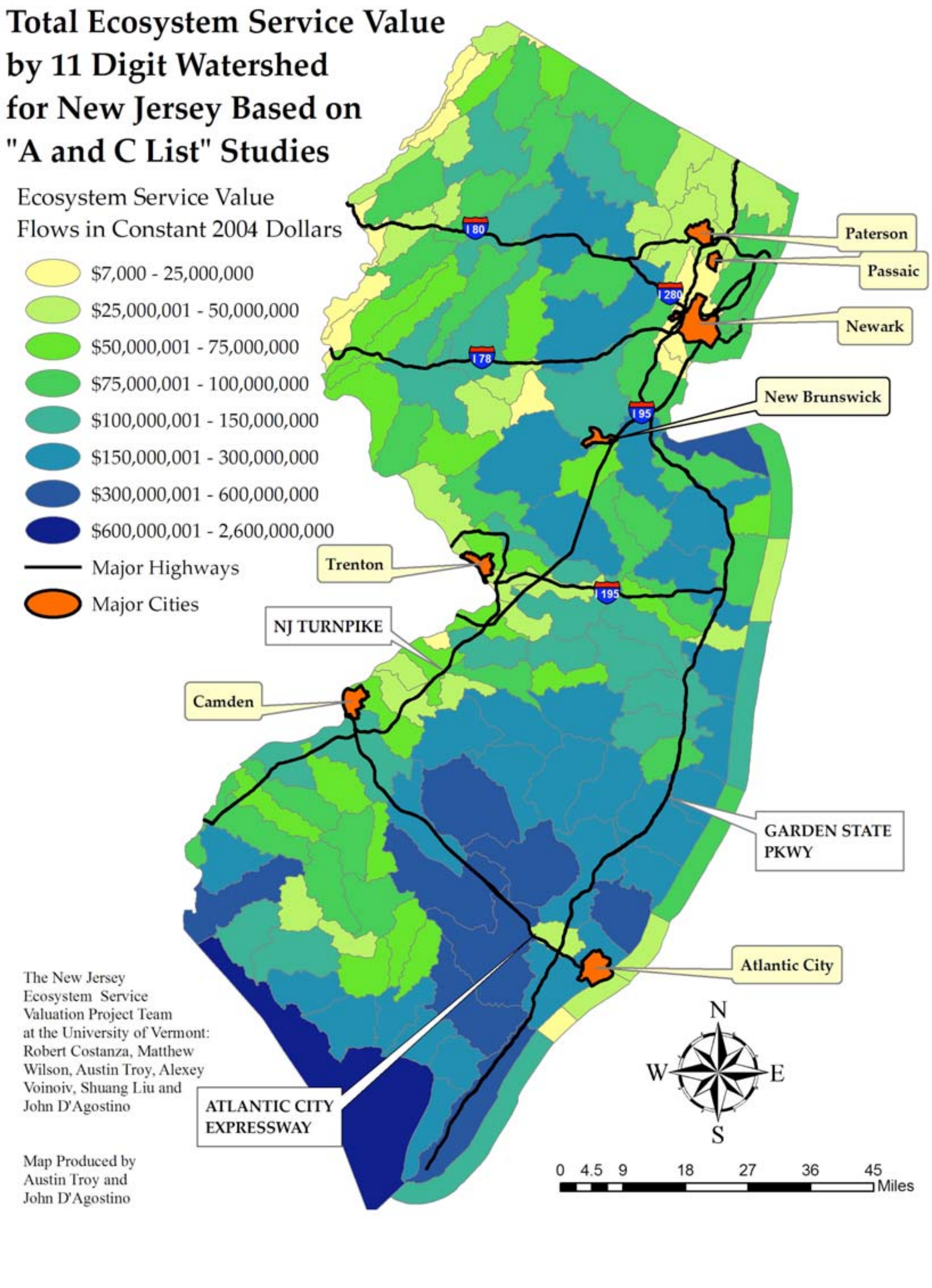


Figure 6: Total Ecosystem Service Value by watershed for New Jersey based on Type A-C studies

Limitations of the Value Transfer Approach

As the previous discussion suggests, not all land cover types generated for the study area could be effectively matched with all possible ecosystem services for each individual land cover type in the study area. This is because the research team's search criteria were focused primarily on Type A economic valuation results, and many landscapes that are of interest from an environmental management perspective simply have not yet been studied for their non-market ecosystem service values. This point is clarified in the following 'gap analysis' tables.

Table 7: Gap Analysis of Valuation Literature (Type A)

	Fresh Wetland	Salt Wetland	Estuary	Open Freshwater	Beac h	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

The data reported in light grey cells in Table 7 show, 163 individual ESV estimates were able to be obtained from 94 peer-reviewed empirical valuation literature for the land cover types included in this study. Areas shaded in white represent situations where we do not anticipate a particular ecosystem service to be provided by a particular land cover type (i.e., pollination by coastal shelf). Areas shaded in dark grey represent cells where we do anticipate a service to be provided by a land cover type, but for which there is currently no empirical research available that satisfies our search criteria.

Table 8: Gap Analysis of Valuation Literature (Type A-C)

	Fresh Wetland	Salt Wetland	Estuary	Open Freshwater	Beac h	Riparian Buffer	Forest	Cropland	Urban Green	Pasture	Coastal Shelf
Gas & climate regulation	1						39		3	3	
Disturbance prevention	1	3	1		2	2					
Water regulation	2								1	1	
Water supply	7		3	5		9	2				3
Soil retention & formation							1			2	
Nutrient regulation			1								1
Waste treatment	1	4					1			1	
Pollination							1	3		1	
Biological control			1				1	1		1	1
Refugium function & wildlife conservation	2	5	6				8	2			
Aesthetic & Recreational	8	3	10	14	4	8	15	2	3	3	
Cultural & Spiritual	1	1	1		1	1	1				1

Total $ Estimates: 210
Total Studies: 100

As the gap analysis for land cover types in Table 8 shows, 210 unique economic valuation data points were identified from 100 empirical sources for use across the cover types generated in this analysis. As the table reveals, by expanding our selection criteria to include synthesis studies (such as Costanza et al. 1997), we were able to fill in several gaps where we anticipated ecosystem services to be delivered by a particular land cover type.

Given the gaps in the available economic valuation data, the results presented in this report 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 (EPA Science Advisory Board Environmental Economics Advisory Committee, 2004), 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.

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.

Value Transfer Conclusions

The results discussed in this section confirm that a substantial and broad range of ESVs ($11 billion to $19 billion) is being delivered annually to New Jersey citizens from a diverse array of land cover types. This variability is consistent with previous findings in the empirical ecosystem services literature (National Research Council 2004). Moreover, each pairing of a land cover type with an ecosystem service presented in Table 4 and Table 5 provides a unique opportunity to "observe" how the "same" service (e.g., disturbance prevention)—when provided by different land cover types (e.g., beach, freshwater wetland, saltwater wetland, estuary)—can vary substantially in its economic value. As the results clearly show, this variability emerges from the valuation literature itself and is not an artefact of any particular study; people appear to value ecosystem services quite differently in different biophysical contexts, and the ESV estimates presented in this report reflect that inherent variability.

In summary, diversity and variability rather than homogeneity and consistency appear to be the best terms for describing the economic values delivered by New Jersey's ecosystems.

Hedonic Approach

In hedonic analysis, the observed sales price of a residential property is statistically disaggregated into a schedule of implicit marginal prices (Griliches, 1971; Quigley, 1970; Rosen, 1974). These unobserved "implicit prices" represent homebuyers' marginal (i.e., incremental) willingness to pay for a property's structural (e.g. lot size, house characteristics), neighborhood (e.g. tax rate, school quality, city service quality) and locational (e.g. proximity to employment, natural amenities and nuisances) characteristics of the property. Sale prices are a better indicator of a property's "true" market value than assessments or appraisals.

Because the attributes are not traded directly in the market and the implicit prices associated with them are not directly observable, they must be statistically derived. A schedule of implicit prices is derived by regressing observed sales price against this set of predictor variables. The resulting coefficients can be interpreted differently depending on the functional form of the model. In the case of a linear model, the coefficients can then be thought of as a marginal change in price due to a one-unit change in that predictor variable, holding all else constant. For the commonly used semi-log model (where price only is log-transformed), coefficients can be interpreted as percentage changes in the response due to a unit change in the predictor. In a trans-log model, that is when both response and predictor are logged, the coefficient on the logged predictor can be interpreted as an elasticity; that is, a percentage change in the response variable due to a percentage change in the predictor. Another way of thinking of this is that in a semilog model, the effect of a marginal change of an attribute on price depends on the price level at which the change is evaluated. In a trans-log model, the effect of a marginal change of an attribute on price depends on both the price level and the attribute level. The interpretation of functional forms is further elaborated upon in Appendix D.

Aggregate welfare benefits associated with a resource which delivers ecosystem services can then be estimated through second stage hedonic regression. This stage is performed far less frequently than the first stage described above due to its technical complexity, myriad assumptions, and data limitations. Second stage analysis quantifies welfare changes resulting from eliminating or creating a resource in question. In the case of this study, we are interested in resources (e.g. forests, wetlands, beaches, etc.) that already exist. So, to value them we must look at how aggregate welfare would change if these resources were eliminated. The value of the forgone benefits is known as opportunity cost. The economics and econometrics behind this stage are highly technical and are described in Appendix D.

A large number of studies have generated valuation estimates for environmental and recreational amenities using hedonic analysis. This includes valuations of open space (Acharya and Bennett, 2001; Irwin, 2002; Riddel, 2001; Riddel, 2002; Smith et. al., 2002), forests (Englin and Mendelsohn, 1991; Garrod and Willis, 1992; Lee, 1997), wetlands (Earnhart, 2001; Mahan et. al., 2000), and water features and associated water quality (Boyle and Kiel, 2001; Carson and Martin, 1990; Gibbs et. al., 2002; Hurley et. al., 1999; Leggett and Bockstael, 2000; Loomis and Feldman, 2003; Steinnes, 1992). Various studies have also used hedonics to study how environmental liabilities are capitalized into housing values as well, such as transmission lines (Harrison, 2002), nuclear power plants (Folland and Hough, 2000), natural hazard zones (Troy and Romm, 2004), heavily polluting manufacturing plants (Anstine, 2003), hazardous waste sites (Michaels and Smith, 1990; Deaton, 2002), nuclear fuel storage sites (Clark and Allison, 1999) and landfills (Hite et. al., 2001).

Market Segmentation

One of the assumptions of the hedonic model is that the results of a model are valid only for a given housing market. That is, the relationship between price and attributes is assumed constant only within a given market segment. For instance, a third bedroom may be worth $800 in suburban Tulsa and $20,000 in suburban Boston, *ceteris paribus* (i.e. adjusting for distance to downtown, school quality, etc). This limits the reliability of hedonic price estimates for value transfer analysis because one housing market may, for instance, value tree canopy cover completely differently from another. However, if it were possible to easily identify and segment housing markets using a standard typology, one could attempt to transfer value estimates from a study site to a policy site located in a similar type of housing market. Unfortunately, studies and methodologies on geographic housing market segmentation are relatively few (Gaubert et. al., 1996; Goodman and Thibodeau, 1998; Goodman and Thibodeau, 2003; Palm, 1978). Therefore, most studies rely on a combination of census data and anecdotal information from people familiar with the housing market to geographically segment markets. In the case of New Jersey, we took such a hybrid approach.

For this project, we purchased data on a sample of 30,000 real estate transactions in central New Jersey for the years 2001 to 2004. This large sample was needed because of the large number of variables being controlled for in hedonic analysis. The more variables there are, there more observations are needed to obtain variation in all of those variables so that the effect of each can be statistically analyzed, independent of the others. The entire state was not analyzed because it would have been cost prohibitive to obtain valid results at that level. Our central New Jersey study area was segmented into seven distinct submarkets: New Brunswick, Princeton, Freehold, Tom's River, Tom's River Fringe, and Barrier Islands Towns (Figure 7). This was done by selecting for the largest urban cores in the study area, resulting in the first five on that list. Surrounding municipal boundaries were then assigned to each one of those five urban cores. Because this resulted in an extremely disproportionate share of the property transactions falling within one market, Tom's River, that market was further segmented into three sub-markets: Tom's River, Tom's River Fringe and Barrier Island Towns.

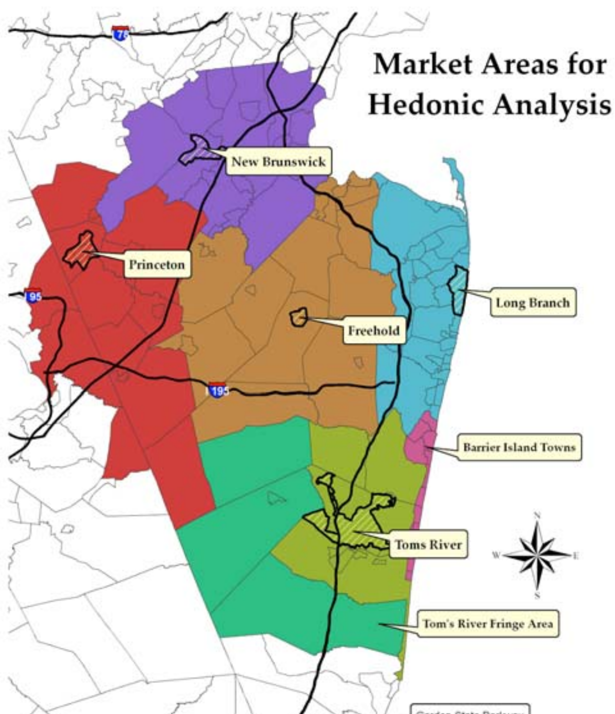


Figure 7: The central New Jersey study area was broken down into seven market areas for hedonic analysis

To ensure that these market areas were generally representative of New Jersey as a whole, comparisons were made based on acreage and percentage of each land cover type, median household income, percent African-American population, percent unemployment, percent with bachelor's degrees, and percent with high school diplomas. Due to time limitations, additional variables were not analyzed. Due to data availability differences, the comparisons were broken somewhat differently for different variables. For land cover, the market area as a whole was compared to New Jersey as a whole. For unemployment, and the two education metrics, values are given for the market area, for New Jersey, and then broken down by each of the four counties included in the market area. For percent African-American and median household income, for which data were more easily available, values are given for the market area, for New Jersey, and then broken down separately by each of the seven sub-markets from the study. Results are given in the three tables below (Tables 9-11).

Table 9: Land Cover Comparison between all of New Jersey and hedonic market area

	New Jersey area	Market area	New Jersey %	Market %
Coastal and Marine				
Coastal Shelf	299,835	-	5	-
Beach	7,837	3,293	0	0
Estuary	455,700	147,228	8	13
Saltwater wetland	190,520	10,392	3	1
Terrestrial				
Forest	1,465,668	222,317	26	20
Pasture/grassland	583,009	87,322	11	8
Cropland	90,455	13,460	2	1
Freshwater wetland	814,479	179,817	15	16
Open freshwater	86,232	11,364	2	1
Riparian buffer	15,146	4,518	0	0
Urban greenspace	169,550	30,023	3	3
Urban or Barren	1,365,742	417,072	25	37
Total	5,544,173	1,126,806	100	100

Table 10: Comparison of education and employment variables between New Jersey and market area with breakdowns by market area county

	% HS diploma	% Bachelor's degree	% Unemployed
Market area	85.1	29.0	4.6
New Jersey	82.1	29.8	4.8
Mercer	81.8	34.0	4.2
Middlesex	84.4	33.0	4.5
Monmouth	87.9	34.6	4.4
Ocean	83.0	19.5	4.9

Table 11: Comparison of income and race variables between New Jersey and market area with breakdowns by seven submarket segments

	Median household income ($)	% African-American
New Jersey	55,146	13.6
Market	60,404	4.9
Princeton	83,364	5.9
New Brunswick	64,111	10.2
Freehold	74,598	4.3
Long Branch	65,269	10.3
Toms River	53,792	2.4
Toms River fringe	54,178	1.6
South Coast	59,071	0.4

The above comparisons document that the market areas selected for analysis are broadly comparable to the rest of the state though there is a small difference (~8%) in the average income between the study area and the state as a whole, and a much more considerable difference in the percentage of African-Americans between the two. Given the large variance in median income both within the state as a whole and within the study area, the relatively small difference in average median incomes should have little effect in biasing results. The race variable, although displaying a greater mean difference is still not a great concern given that within the study area there are two market segments, Long Branch and New Brunswick, near the state average. Given that, in the design phase of the hedonic study, the primary objective to sample contiguous areas with a considerable presence and diversity of natural amenities and landscapes, and that the study design did not attempt to achieve representation of the state relative to income and race, these relatively small differences are to be expected. While it may be difficult to generalize these results to certain areas of the state, such as those with low incomes, high minority levels, or low levels of natural amenities, results can still be generalized to a considerable portion of the state.

Hedonic Methods

In a hedonic analysis, the observed dependent variable is statistically disaggregated into implicit marginal prices for each explanatory attribute. For the New Jersey case study, this was done by regressing property sales price against a set of independent variables describing the lot, neighborhood, socio-economic characteristics, location and environmental amenities. So defined, the hedonic pricing equation is expressed:

$$Ln(P_{it}) = \iota_0 + \iota_1 L_i + \iota_2 N_i + \iota_3 S_i + \iota_4 T_i + \iota_5 E_i + \iota_{it} \qquad (2)$$

Where: P_{it} = Sales price of house i at the time of transaction t

ι_0 = intercept (note: this term has little significance to the results)

$\iota_{1\ldots n}$ = vectors of regression coefficients

L_i = vector of lot/structural characteristics of house i

N_i = vector of neighborhood characteristics of house i

S_i = vector of socio-economic characteristics of house i

T_i = dummy variable indicating the year in which house i was transacted

E_i = vector of environmental characteristics of house i

ι_ι = regression error term

As expected, the component variables of each vector differed by submarket because of differing relationships between price and attributes by market. A list of all variable names with descriptions is given in Table 12 below. The specific variables used in each model can be seen by examining the model results in Appendix E Tables 1-7.

Table 12: Variable Names and Descriptions

Variable Name	Description
SalePrce	Residential Property Transaction Price ($)
Lot variables	
Liv.Area	Living Area (sq. feet)
PropTax	Assessed Property Taxes ($/year of transaction)
Imp.Val	Structural Improvement Value of the Property ($)

LotAcres	Property Area (acres)
two.story	Dummy Variable for Two-Story Homes (1/0)
al.siding	Dummy Variable for Homes with Aluminum Siding (1/0)
NEW	Dummy Variable for Home Construction within years 1994-2004 (1/0)
OLD	Dummy Variable for Home Construction > 75 Years Ago (1/0)
House.age	Continuous Variable for Year of Home Construction
Neighborhood (ft.)	
D2AIRPRT	Distance to Nearest Airport
D2URBAN	Distance to Nearest NJDEP-designated Urban Area
D2RETAIL	Distance to Nearest Major Retail Center
D2CLUB	Distance to Nearest Country Club/Golf Course
D2TERMNL	Distance to Nearest Transportation Terminal (Bus Depot, Train Station, etc.)
D2CONTAM	Distance to Nearest Contaminated Site
D2HIX	Distance to Nearest Highway Exit
Socio-economic	
P.VAC	Percentage of Vacant Homes
MED.HH.INC	Median Household Income ($)
P.OWN.OCC	Percentage of Owner Occupied Homes
P.BLK	Percentage of Population of African-American Ethnicity
P.HISP	Percentage of Population of Hispanic Ethnicity
Transaction Date	
X2002	Transaction Occurred in 2002
X2003	" " " 2003
X2004	" " " 2004
Environmental Amenities	
ENV.SENS	Dummy Variable for Property Location within Environmentally Sensitive Region (1/0)
D2UN.WET	Distance to Nearest Unprotected Wetland Area (ft.)
D2UN.FOR	Distance to Nearest Unprotected Forest Area (ft.)
D2WATER	Distance to Nearest Significant Body of Water (ft.)
WATER100	Dummy Variable for Property Location within 100 ft. of Significant Water Body (1/0)
D2SPARK	Distance to Nearest Small Park (< 50 acres)
D2MPARK	Distance to Nearest Medium Park (50 - 2000 acres)
D2LPARK	Distance to Nearest Large Park (> 2000 acres)
D2BEACH	Distance to Nearest Beach (feet)
BEACH1	Dummy Variable for Property Location within 300 ft. of Nearest Beach (1/0)
BEACH2	Dummy Variable for Property Location between 300 and 2000 ft. of Nearest Beach (1/0)
FLOOD.SFHA	Dummy Variable for Property Location within FEMA Special Flood Hazard Area (1/0)

One of the notable variables not included in any of the regressions was school quality. Inclusion of this factor was attempted for five of the seven markets by obtaining elementary school district proficiency averages (percent partially proficient, percent proficient, percent advanced proficient) from the New Jersey Department of Education, assigning them to municipal elementary school districts (roughly the size of individual municipalities), and including them as independent variables. While these variables were significant in most cases, they had only a minute effect on overall model fit (R-squared ~.002) and appeared to make some of the main effects results unstable; they caused some of the coefficients on environmental amenities to lose their significance. Hence, these variables were dropped. It is recommended that any future attempt to model housing markets in the state break down school quality by boundaries for individual elementary schools, rather than elementary school districts, because of the extreme variation that is generally found within a district. Unfortunately, such data was not digitally available for the study area at the time of this study. It should also be taken into account that at the coarse level of entire school districts, school quality was largely being proxied by control variables such as median household income, or percent home ownership. For that reason, adding school quality as an additional independent variable would, as noted above, have little impact on the overall explanatory power of the regression equations.

The details of the regression analysis are very complex and are described further in Appendix E. The main environmental variables found to affect housing prices included distance to small (<50 acres), medium (50-2000 acres), and large parks (2000+ acres), location adjacent to (0-300 feet) or near (300-2000 feet) the beach, distance to the nearest beach, distance to water bodies, adjacency to water bodies (<100 feet), and location within "environmentally sensitive zones" as designated by the New Jersey Office of Smart Growth. This includes large contiguous areas of ecological significance, including critical water supply sources, habitat areas, trout streams, scenic greenbelts, wetlands, etc. An additional variable for distance to unprotected forestland was also included, but because it was either insignificant or had the opposite sign of expected (i.e. forest proximity decreases home value) for different markets, it was excluded. Another main effects variable that was attempted and dropped was distance to nearest wetland. As described in the Appendix, the regression analysis yields a set of statistical coefficients, quantifying the relationship between price and each attribute. It should be kept in mind that because these models used a semi-log functional form (where price is log transformed), that means that each coefficient (given in Appendix E) describes a percentage in price due to a change in the attribute level. This is because in the semi-log specification, the effect of an attribute change on price depends on the price at which it is being evaluated. In most models, some predictors are also log transformed. Where this is the case, the coefficient can be interpreted as elasticity, meaning that the increase in price due to an increase in an attribute depends on both the price and attribute level at which it is evaluated.

Data

A data set covering more than 30,000 residential property transactions from Mercer, Middlesex, Monmouth, and Ocean counties between January 2001 and August 2004 was obtained from First American Real Estate Solutions' RealQuest database. Attributes included sale value ($), calculated property tax ($), total living space area (sq. feet), property improvement value ($), lot acreage, transaction date, property street name, town and zip code, etc. Properties for which necessary lot attributes were absent were excluded. The real estate set was then address geocoded in ArcMap, using a detailed streets layer containing address range information from Geographic Data Technology (GDT) Inc. This yielded a GIS layer showing a point for the location of each transaction. The geocoded transactions were then examined for missing or flawed attribution; systematically flawed attributes were corrected when possible while properties for which sale value, transaction date, or similarly necessary analysis characteristics was not provided were exempted from the final transaction set, yielding a final set of 27,733 central New Jersey transactions for the January 2001 – August 2004 period.

Spatial data was obtained from the New Jersey Department of Environmental Protection's Bureau, including land cover by watershed, shore type, state water bodies and wetlands, census block groups, contaminated sites, state and local parks. Other data layers were obtained from the New Jersey Office of Smart Growth, including urban core boundaries, sewer service area boundaries, and environmentally sensitive areas, and from GDT, including transportation terminals, major retail centers, country clubs, and airports.

A number of locational variables were attributed to each transaction point using the "spatial join" function in ArcGIS (ESRI, Inc.). This was done to calculate distances to both amenities and disamenities, including nearest small, medium, and large protected areas (e.g., parks, conserved lands); unprotected wetlands and forests; water bodies; environmentally sensitive areas; contaminated areas; country clubs; airports; transportation terminals; major retail centers; defined urban areas; major highways, and numerous others. Overlay analysis was used to determine whether each house was located within a number of environmentally-relevant zones, including high-risk FEMA-delineated flood zone, the Coastal Areas Facility Review Act (CAFRA) zone, the two beach proximity zones, and the water proximity zone. Based on work by Troy & Romm (2004), the beach variable was defined in two ways: for each property, the distance to the nearest beach was calculated in addition to two dummy variables designating property locations within 300 feet and between 300 and 2000 feet from the beach ecosystem. This accounts for both an adjacency effect as well as a distance to access effect. All continuous distance variables were measured in feet and variables indicating property location within a zone were given as a binary variable (i.e., a variable which can only take on the values zero or one).

Hedonic Analysis - Results

Overall, our hedonic models had strong R-squared values ranging between 0.70 and 0.87. An R-squared value of 1.0 would mean that the regression equation was able to account for 100% of the variation in the dependent variable, e.g. housing price. The models were constructed such that almost all included control variables were significant and of the expected algebraic sign (positive or negative). Complete results with all coefficients and test statistics for each market are given in Appendix E.

Not all environmental amenity variables were significant or of the expected sign. Table 13 below, which shows all main effects variables for all markets, highlights in grey all variables that have the opposite of expected sign. It also shows with NA all those variables which were not significant or not applicable. The following are major results:

The variable for distance to large parks has the correct sign and significance for three markets. For another three, large parks are valued negatively and for one they are not statistically significant. Small parks are statistically significant with the correct sign in five markets. Medium parks have the correct sign and are statistically significant for one market only. In the pooled model, where all markets are regressed simultaneously, only small parks have the correct sign and are statistically significant. Moreover, both the variable on distance to small parks and acreage of nearest small park have the correct sign.

The Beach 1 zone is significant and of expected sign for each market where there is a beach, while the Beach 2 zone has the correct sign and is significant for two of the three markets where it is applicable.

The variable for proximity to water bodies is significant and of the correct sign for two markets, and not significant in the others. The dummy variable for water proximity zones is significant in only one market, where it has the correct sign.

Environmentally sensitive zone. The environmentally sensitive zone dummy variable is significant for two markets, for which it has the correct sign for both.

Distance to Beach. Finally, the distance to beach variable, which was only significant in two markets (because the zonal dummy variable tended to be a better predictor), was of expected sign in both.

Table 13 also gives price differentials for environmental amenities showing how average price, holding all else constant, increases or decreases with proximity or adjacency to an environmental amenity. This was completed by solving the hedonic equation for each market, holding all control variables at their mean values, while varying the distance to an amenity, and then comparing the change in price due to that location shift. For zonal dummy variables (1/0), comparisons were given by solving the equation for a property both in and out of the zone and comparing the results.

Table 13: Main Effects Variable Differentials by Market

Continuous Variables			Market and Price Effect in 2004 USD						
Resource	**Moving From:**	**Moving To:**	**Princeton**	**New Brunswick**	**Freehold**	**Long Branch**	**Tom's River**	**Tom's River Fringe**	**South Coast**
Beach	1000 ft	100 ft	743	458	NA	NA	NA	NA	NA
Beach	1 mile	100 ft	4,260	2,623	NA	NA	NA	NA	NA
Beach	5 mile	100 ft	21,249	13,068	NA	NA	NA	NA	NA
Water	1000 ft	100 ft	(24,991)	NA	NA	NA	13,952	NA	40,847
Water	1 mile	100 ft	(44,121)	NA	NA	NA	23,556	NA	67,843
Water	5 mile	100 ft	(63,527)	NA	NA	NA	32,483	NA	92,104
Small Park	1000 ft	100 ft	NA	21,,006	617	2,216	NA	(2,391)	7,489
Small Park	1 mile	100 ft	NA	35,240	3,533	12,562	NA	(14,251)	32,861
Small Park	5 mile	100 ft	NA	48,297	17,539	59,291	NA	(86,417)	178,541
Medium Park	1000 ft	100 ft	2,450	(738)	(929)	NA	309	(7,643)	(6,491)
Medium Park	1 mile	100 ft	13, 905	(4,278)	(5,386)	NA	1,775	(13,381)	(39,603)
Medium Park	5 mile	100 ft	66,001	(22,492)	(28,339)	NA	8,875	(19,110)	(272,485)
Large Park	1000 ft	100 ft	1,437	(10,132)	(1,318)	1,486	NA	14,922	(4,458)
Large Park	1 mile	100 ft	8,202	(17,728)	(7,672)	8,478	NA	25,054	(26,935)
Large Park	5 mile	100 ft	40,023	(25,300)	(41,163)	32,804	NA	34,363	(175,763)
Dummy variables	**Zone size**								
Environmentally sensitive zone	Defined by map		NA	NA	NA	34,525	8,562	NA	NA
Beach 1 zone	0-300 ft from beach		NA	NA	81,202	99,574	194,066	NA	100,169
Beach 2 zone	300-2000 ft from beach		NA	NA	(28,397)	15,900	NA	NA	44,107
Water zone	100 ft from water		NA	NA	NA	32,912	NA	NA	

NA= not applicable or not significant
Expected sign
Opposite of Expected Sign

Unsurprisingly, the results indicate that beaches are very highly valued by the property market. In the case of Tom's River, being adjacent to a beach increases property value by almost $200,000. Being within 2000 feet of a beach can also increase property value by over $40,000, holding all else constant. Water body proximity seems to be positively valued. Where significant, environmentally sensitive zones appear to be positively valued, although it is hard to interpret what that means, since this zone includes so many diverse landscape types. Nevertheless, that result indicates that areas of ecological significance are, in general valued positively. It may also indicate that people value the fact that the environmental sensitive zone designation limits future development opportunities in the area and gives some assurances of continued future integrity. As mentioned in the methods section above, otherwise unprotected natural lands, including forests and wetlands, receive no positive valuation at all, indicating that *natural landscapes are not highly valued if they are subject to potential future development*.

Finally, perhaps the most equivocal results relate to protected parks and open space. While we expected to find a positive valuation for all open space in all markets, the actual results were extremely variable. Overall, small parks tend to be the most highly valued, perhaps because they are seen by homebuyers as representing a compromise between urban access and rural or suburban amenities. Medium and large parks can be either positively or negatively valued, depending on the market. The differences may depend on a number of factors. First, large and medium park proximity may be proxying something else, since it is unlikely that residents would negatively value parks in and of themselves. We tried to control for distance to urban areas, highways, and urban amenities. But these efforts are still not sufficient to adequately control for all the locational factors that draw people towards cities. Most notably, we did not have the time or resources to develop a robust indicator of access to employment opportunities, and the quality of those opportunities which is one of the most important determinants of housing price. Hence, where large and medium parks are valued negatively, this may be because those variables are proxying "ruralness" or low levels of economic development, even despite the use of similar control variables. In cases where those variables are the correct sign, it is probably because the control variables are more adequate for those markets.

Another difference may be due to differences in the preferences of homeowners within a given market. For instance, the Princeton market, which has the highest income of any of the markets, also is the only market that positively values both large and medium parks. Hence, the degree or sign of valuation may relate to socio-economic differences that inform preferences. Next, it may relate to the abundance of open space. In areas where open space is already abundant, proximity to protected open space has relatively little value, since almost any house in the area will have functionally similar open space access. Finally, the difference might be due to differences in the characteristics of the parks themselves. Some parks may be well kept and others not. Some parks may be dominated more by impervious surface and non-natural features and other not. Some parks may have high crime or be associated with other problems. In small parks, these differences are more likely to average out because of the large number of them.

Second Stage: Hedonic per Acre Value Estimates

In theory, the first stage hedonic results can be used to derive per acre aesthetic amenity and recreational ecosystem service values. Due to the time consuming nature of this undertaking, we conducted a preliminary second stage analysis for some of the relevant land covers.

A number of serious technical challenges have confronted us in attempting this stage. For instance, we found that households valued proximity to protected areas—not just natural cover types. Unfortunately, "protected areas" is not an ecological category that occurs in our ecosystem service valuation typology. An analysis of land cover in protected areas finds that the cover types in our study area tend to be quite diverse within the park boundaries, including forests, urban green space, open water, riparian zones, and considerable areas of wetlands. Hence, when looking at how a set of households value a given park, we are really looking at a composite of cover types, which makes it difficult to value any one park in a definitive fashion. This is complicated by the fact that valuation estimates vary by real estate market, many markets appear not to value them, or value them negatively, probably due to co-linearity in the models, and many big parks are on the boundaries between two or more markets, so that valuing such parks becomes almost analytically intractable. This would not have been the case had we found positive values for generic unprotected forests or wetlands, since these types are common throughout the state and do not have to be part of a larger object (e.g. park) to be considered.

While we could not estimate per acre values for all cover types in this study, we were able to derive them for the urban green space and beach types using second stage methods. Because the beach coefficients that were analyzed for the second stage were binary or "dummy" variables, the methodology was far simpler and only required adding up price differentials. However, the green space analysis, based on first stage analysis of small parks, involved continuous variables and was far more involved. The former is described first.

Using coefficients for the two beach zone dummy variables from the South Coast market and the Long Beach market we estimated the amount that each house in the data set increased in value due to proximity to the beach. To account for the fact that there are many more houses than appear in the data sample, we used census block group data to determine the ratio of the count of actual household units to sample households in each block group. This number was then multiplied by the occupancy rate for each block group to eliminate buildings that are either permanently vacant or that are vacant for much of the year. The resulting multipliers were then assigned back to sample houses. The total value of the beach in this market was then estimated by multiplying the value increase per sample house due to beach proximity by the household ratio multiplier, which varied by block group. These were then summed and divided by the acreage of beach in each market to obtain a stock value. Yearly flows were obtained by multiplying the resulting number by a 3% discount rate. The results indicated a yearly amenity value flow of $43,718/acre in the Long Branch market and $31,540/acre in South Coast market, both of which are relatively close to the transferred value used in this study of $42,147. At a 5% discount rate, these numbers go up to $72,864/ acre and $52,567/acre respectively, which are higher than the transferred value. Results are described in Table 14.

Table 14: Estimated per acre stocks and flows of urban greenspace and beaches based on first and second stage hedonic methodology

	Urban Greenspace-acreage method	Urban Greenspace-distance method	Beach Long Branch	Beach-South Coast
Stock	$914,000,000	$1,010,000,000	$910,797,000	$440,512,813
Acres	2,738	2,738	625	419
Per acre stock value	$333,820	$368,882	$1,457,275	$1,051,343
Per acre flow (3%)	$10,015	$11,066	$43,718	$31,540
Per acre flow (5%)	$16,691	$18,444	$72,864	$52,567

Urban green space values were determined by looking at small parks (less than 50 acres) within the hedonic study area. While not all of them fell within designated "urban cores," by overlaying sample parks on aerial photos within GIS software, it could be seen that almost all are in fairly urban or heavily suburbanized settings, making it reasonable assume that these small parks are functionally representative of an "urban greenspace" category as used in ecological economics valuation literature. Two different approaches were tried. In the first, the second stage hedonic methods described in the Methods section were applied to the distance to nearest small park variable. For the five markets where small park distance had the expected sign and was significant (Princeton, Freehold, Long Branch, New Brunswick and South Coast), the partial derivative of price with respect to small park distance was taken for each model (including, trans-log, semi-log and quadratic models). The resulting equation was then solved for every observation to give the shadow price. Observations from all five markets were combined in a spreadsheet and shadow price was regressed against the distance to nearest small park variable and median household income (as a demand shifter) for the pooled data. The pooling of all data served the purposes of avoiding the identification problem described in the Methods. The functional form used for this regression was semi-log. The coefficients of this equation were then used to derive the inverse demand curve, which was integrated at the mean distance value, aggregated by households, and divided by park acreage.

The second method attempted to look at the park size attribute to estimate welfare measures, rather than the distance attribute, in order to triangulate results from the latter. In many of the market-specific hedonic models, the park size variable was not significant. However, when data from all markets were pooled and regressed together for first stage hedonic analysis, this yielded a significant coefficient with the correct sign on the park size variable. Because of the identification problem that would have resulted in doing a second stage regression with pooled data, in this case we instead solved each observation for the contribution of park acreage to its price and then aggregated using the multipliers described above, again dividing by total small park area.

As we had hoped, the numbers were extremely similar for the two methods, and were considerably higher than the transferred values of $2,473 per acre for urban greenspace. Using a 3% discount rate the park distance method yielded a yearly flow of $11,066 per acre and the park area method yielded a yearly flow of $10,014 per acre. At 5%, these become $18,444 and $16,691 per acre. These are also given in Table 14.

Ecosystem Modeling Approach and Results

The ecosystem valuation methods described earlier in this report have been criticized on various grounds.

- The value transfer approach is sometimes criticized because it uses values for the "average" ecosystem of a given type, e.g., wetlands; since every ecosystem of a given type is unique in some respects, it is argued that average values cannot capture that uniqueness.
- The hedonic value approach relies on the assumption that consumer perceptions of differences in environmental quality are reflected in housing prices; however, several important ecosystem services are not perceived directly by humans and therefore presumably will not show up in hedonic prices.

The services which cannot be directly perceived include climate regulation, disturbance prevention, freshwater regulation and supply, and waste and nutrient regulation. These services are directly connected to an ecosystem's primary production, nutrient dynamics, and hydrology; these ecosystem "functions" in turn directly affect the quality and quantity of services provided by the ecosystem. Ecosystem functioning is driven by such factors as land use, geology, species mix, etc.

Modeling Approach

Scientists who perform ecosystem valuation studies are beginning to develop techniques to assess the impacts of some of the many relevant site-specific factors on the quantity and quality of ecosystem functions and services. As part of this project, we undertook one such type of analysis called spatial modeling. In essence, this technique uses complex computer software to model the physical interactions of ecosystems and human communities in a given landscape in a dynamic mode (i.e., a mode in which the physical state of the landscape components at a given point in time directly determines the physical state of the landscape at subsequent moments).

The specific software we used is a spatially explicit, process-based model previously developed to integrate data and knowledge over several spatial, temporal and complexity scales and to aid in regional ecosystem and land use management (Costanza et al., 2002). The model 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. The spatial resolution is variable, with a maximum of 200m x 200m to allow adequate depiction of the pattern of ecosystems and human settlement on the landscape. The temporal resolution is different for various components of the model, ranging from hourly time steps in the hydrologic sector to yearly time steps in the economic land use transition module.

The model just described is capable of general application, and has been calibrated for the Patuxent River watershed in Maryland, several of its subwatersheds, including Hunting Creek subwatershed, and a few watersheds in Vermont. In this context, calibration refers to determining the numerical constants in a given mathematical relationship, which is used in the model to describe certain processes. Since there are always uncertainties involved in choosing the right formalism and comparing the model results with data, which are also uncertain, calibration is used to improve model accuracy and to incorporate some of specific features of a landscape or a particular case study, which could not be picked by the choice of processes and their formalizations. A model thus calibrated for one watershed cannot be directly applied to another watershed without extensive and expensive recalibration based on local data for the new watershed and its component subwatersheds. We can nonetheless use a model calibrated to a particular watershed in another state to derive non-quantitative relationships applicable to New Jersey. We do this by "exercising" or experimenting with the existing model to create spatial pattern and context-based relationships of a type that could be applied in New Jersey, and could be certainly improved if and when the model is recalibrated with New Jersey parameter values based on New Jersey data. (The analytic methods used in these experiments are extremely complex and are described in detail in Appendix D.) Since Maryland and New Jersey are geographically and climatically very similar, such transfers of non-quantitative model results can safely be incorporated in the present study.

For example, using the model, we could create wetlands or forests of varying patch size and observe how freshwater regulation and supply services vary in the model watershed. We can thus identify the type of relationship (e.g., linear, exponential, etc.) that exists between patch size and freshwater regulation and supply services. We performed this type of modeling experiment for several of the ecoservices mentioned above, varying the assumptions on the spatial patterns of land use within the model watershed. Although it is beyond the scope of the current project, the resulting set of functional relationships could be built into a GIS-based system to allow values for these services to be adjusted to take New Jersey-specific spatial effects into account.

Determinants of Ecosystem Services and Functions

The level and economic value of ecosystem services of a given type provided by a given ecosystem depends on a variety of factors, including the ecosystem's size, "health," and location relative to human communities and other ecosystems. At the extreme, each specific ecosystem, e.g., each patch of forest or wetland, would need to be evaluated on its own to assign it a value. This degree of detail is impractical for a region as large as New Jersey, which is why the transferred value analysis used the average of values from prior studies.

Running a model of this type provides an opportunity to quantify several indicators related to ecosystem functioning and ecosystem services. In particular, the model we used for the project can track two variables that are related to certain important ecosystem services.

- Concentration of nutrients (in this case nitrogen) is an important indicator of water quality and how the watershed performs towards amelioration of water pollution.
- Net Primary Productivity (NPP) has been suggested in several studies (cf. Costanza et al. 1998) as a proxy for total ecosystem services value. NPP describes how fast the vegetation grows and therefore is an indicator of the existing amount of vegetation and its health.

The model we built includes variables that can tell how these indicators may change under different scenarios of land use, climate, and other changes in space and time. We can "exercise" our model, e.g., run it under various climatic conditions, and "drive" it by changing patterns of land use to help us better understand how such factors impact the ecosystem services under consideration. To help us understand how allocation of land use affects the two proxies for ecosystem services described above, we first ran 17 scenarios for the model watershed, grouped into a baseline and three sets of experiments as follows:

- Baseline: one scenario representing existing land uses in the model watershed as of 1990, and one representing a hypothetical baseline with the entire watershed forested (Figure 8).
- Extent of land conversion: seven scenarios varying the percentage of forest converted to agriculture from 15% to 100% (Figure 8).
- Location of land conversion: two scenarios converting 30% of forest to agriculture but varying the location of the preserved forest between uplands and lowlands (Figure 9).
- Buffers: six scenarios varying the nature (agriculture or forest), size (one, two or three cells), and location (lowland or midland) of stream buffers in the watershed (Figure 9).

We have been focusing on the conversion between forested and agricultural land uses, while similar experiments could be conducted for other types of landuse change, say conversions from forest to residential. Once again, this choice was primarily driven by the existing data and the confidence in model performance, which was the highest for these two land use types in Hunting Creek watershed. In Table 15 we present an overview of these scenarios, as well as the values for NPP and Total Nitrogen Runoff generated for each scenario. The scenarios are also depicted in Figures 8 and 9 in map form; green represents forest, yellow represents agriculture, and blue represents streams. Gray and black cells stand for low and high density residential, respectively; these land uses were fixed in space in our experiments.

Table 15: Summary of Scenarios Analyzed with the Ecosystem Model

Map Code	# cells	N total (mg/m)	N station (mg/m)	NPP ($kg/m^2/y$)	Scenario description
LU90	1172	1228	208	2.58	1990 land use for model watershed
LU_F	1653	972	154	3.48	Entire model watershed forested
LU_F15	1389	1028	180	2.95	About 15% of forest randomly converted to agriculture
LU_F20	1334	1048	185	2.83	About 20% of forest randomly converted to agriculture
LU_F30	1132	1092	289	2.41	About 30% of forest randomly converted to agriculture
LU_F35	1041	1178	280	2.23	About 35% of forest randomly converted to agriculture
LU_F50	807	1207	264	1.74	About 50% of forest randomly converted to agriculture
LU_F50a	838	1241	339	1.82	About 50% of forest randomly converted to agriculture (an alternative trial)
LU_Agro	0	1493	443	0.00	All forest converted to agriculture
LU_F30a	1132	1393	307	2.48	Forest preserved on upland; while about 30% total converted to agriculture on lowland
LU_F30b	1132	990	287	2.38	Forest preserved on lowland; while about 30% total converted to agriculture on upland
LU_F30ha	1132	1079	213	2.35	Small agricultural buffer
LU_F30hf	1132	1100	272	2.41	Small forest buffer
LU_F30hfm	1132	1037	274	2.41	Medium forest buffer
LU_F30hgbig	1132	1096	288	2.40	Large forest buffer
LU_F30hfbiglow	1132	1084	290	2.40	Large forest buffer, lowland priority
LU_F30hfbigmid	1132	1015	287	2.39	Large forest buffer, midland priority

Key:
mg/m = milligrams of nitrogen per meter of water column in stream
$kg/m^2/y$ = kilograms of NPP per square meter per year
N station = nitrogen mg/m at the mid-watershed gauging station

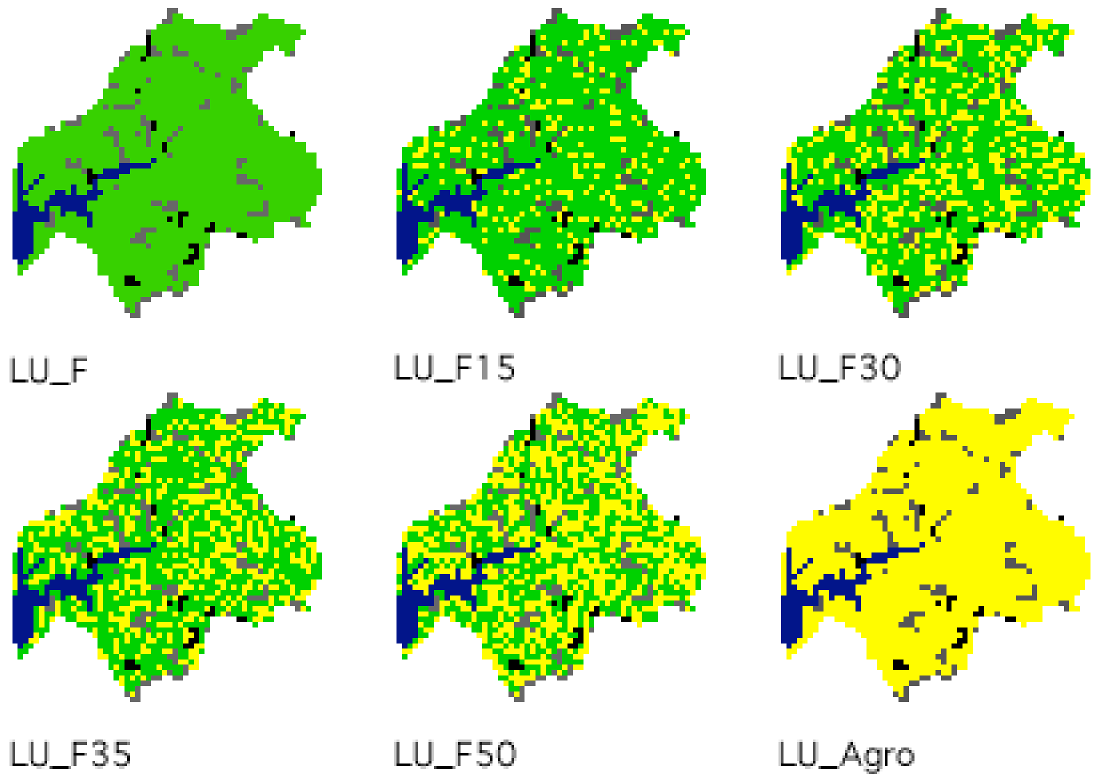


Figure 8: Scenarios for analysis of spatial allocation change

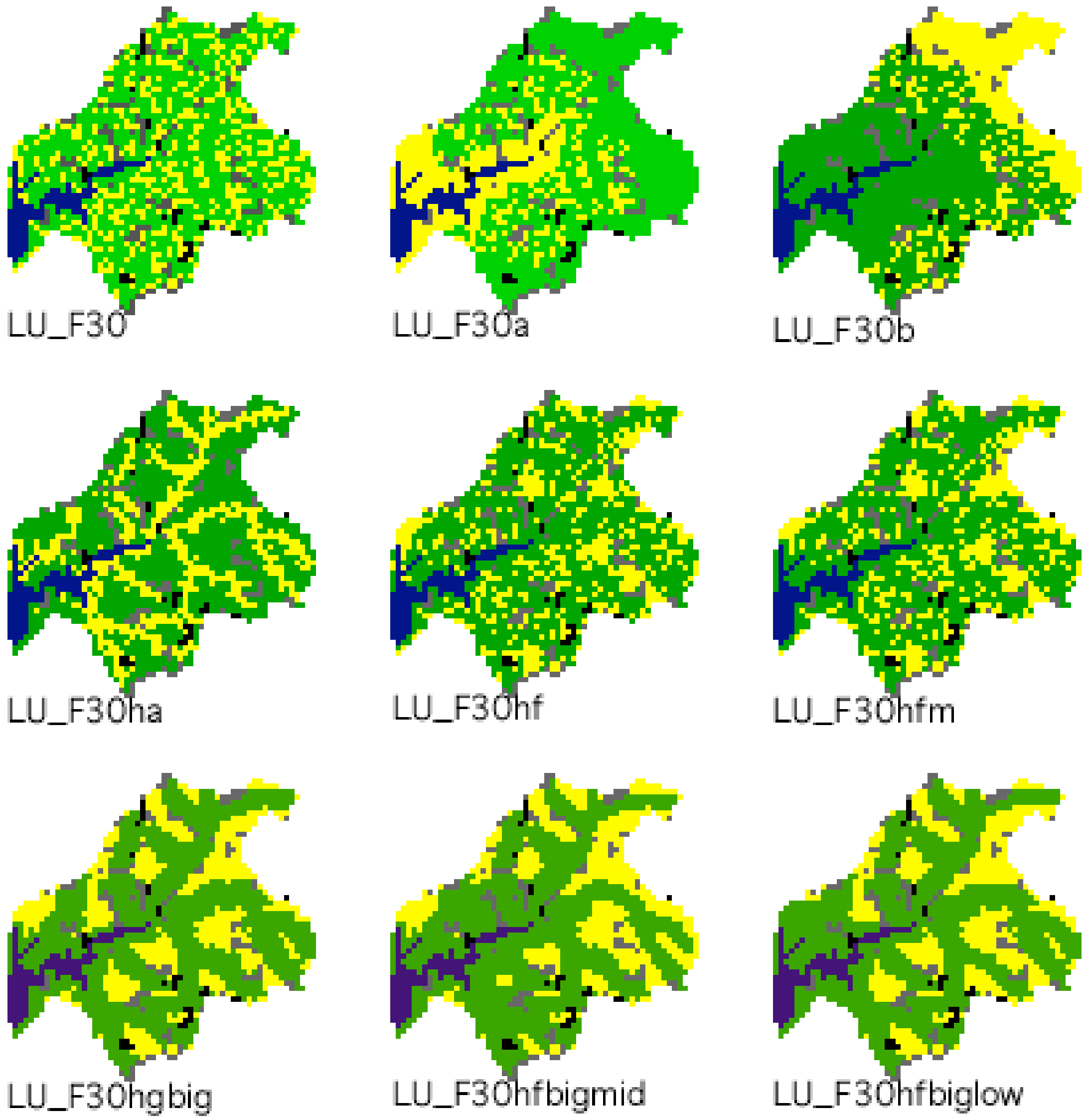


Figure 9: Scenarios for analysis of spatial pattern change

Net Primary Production (NPP)

As noted earlier, net primary production (NPP), excluding agriculture and urban areas, can be treated as an indicator of ecosystem health and ecosystem service levels (Costanza et al., 1998). NPP in the Hunting Creek watershed is primarily provided by the forested areas. Different land use patterns result in quite significant spatial variations in NPP; however, total NPP for the watershed does not seem to be related to the spatial patterns and is almost entirely driven by the total number of cells in the forest land use type (Figure 10). The small variations in NPP that we see in Figure 10 and Table 15 are caused by slight differences in the factors that determine nutrient and water supply in scenarios where different spatial allocations of a constant number of forested cells are assumed.

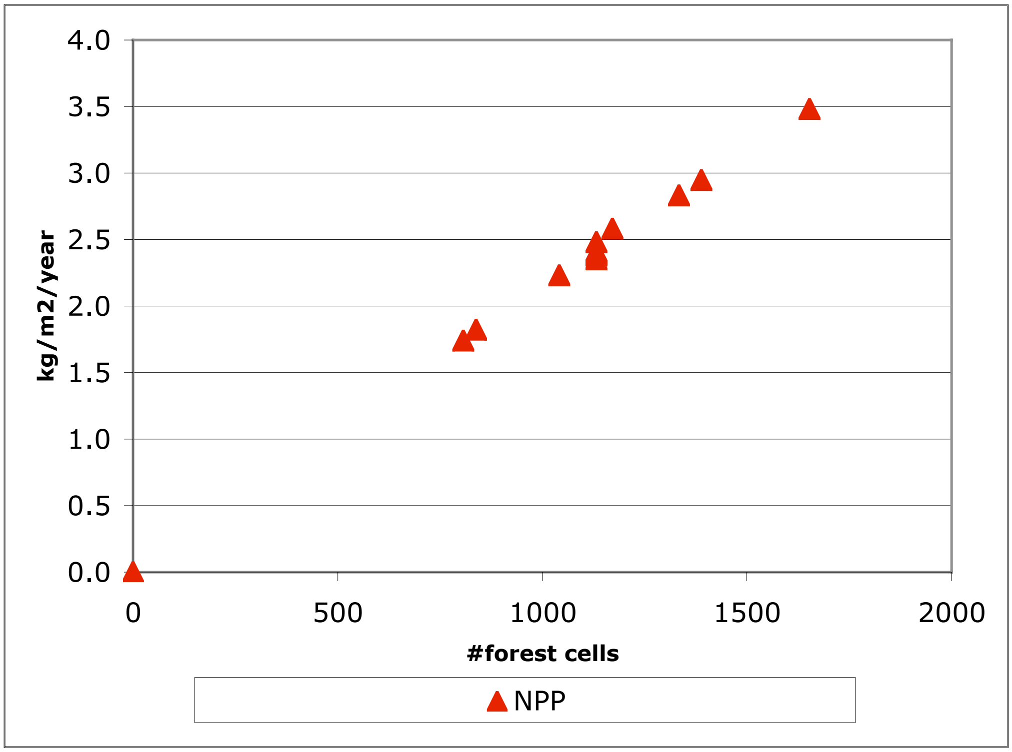


Figure 10: Total NPP as a function of forested cells in the watershed.

We can conclude from this that in terms of NPP, the precise spatial patterns are not very important. There might be more spatial variance if the changes in spatial allocation resulted in larger changes in the suitability of the landscape for plant growth, in which case water or nutrient limitations might result in more dramatic variations in the NPP index.

Nutrient Loading

The next indicator we tracked is nutrient runoff as a function of the spatial distribution of various land use types. This indicator serves as a measure of the quantity and quality of an ecosystem's water regulation services.

For scenarios that vary only the number of cells that are forested, we obtained a response that is very close to linear (Figure 11). In other words, the more forested cells in the watershed, the lower the amount of nutrients (nitrogen in this case) delivered to the estuary. If the spatial distribution of cells is random each time, the response is again almost exactly linear. However, if there are non-random patterns in the arrangement of cells of a particular type, such as what we see for the existing land use pattern (LU90), or in some of the special cases considered below, we see some deviations from the linear relationship (see the outlier point in Figure 11).

These deviations become more obvious if we run the model through the group of scenarios that have the spatial pattern of forests changed as shown in Figure 9. In this case we observe quite substantial (almost 50%) variations in the water regulation services provided, even though the overall proportions of various land use types in the watershed remain constant (Figure 12A). It should be noted here that we have used the same number (1,132) of forested cells in all of these particular model runs, and it is only how we distributed those forested cells across the landscape that was changed.

Figure 12B presents the same spatial changes but reports results for the gauging station that is located in the middle of the stream in the model watershed and mainly covers the upper left (northwest) corner of the watershed. The variations are quite substantial and do not relate well to the changes that we see in Figure 12A; different patterns of land use have different effects on the watershed as a whole and on the sub-watersheds. In other words, there is significant spatial heterogeneity in terms of the water regulation services provided.

The difference in results between Figures 12A and 12B highlight the relevance of policy and regulatory objectives to ecosystem analysis. If we are interested only in the water quality in the estuary zone, then the entire watershed can be treated as a single unit and we need not be concerned about the variations of forest distribution among the different parts of the basin. However, if we are concerned about stream health throughout the watershed, then the spatial gradients in nutrient levels become highly important, and we have to take into account the fact that land use change in one area will impact adjacent areas downstream. In other words, the estuary at the bottom (southern) end of the watershed may experience a very different level of disturbance from upstream portions of the river due to various factors, e.g., dilution of nutrients as they flow downstream.

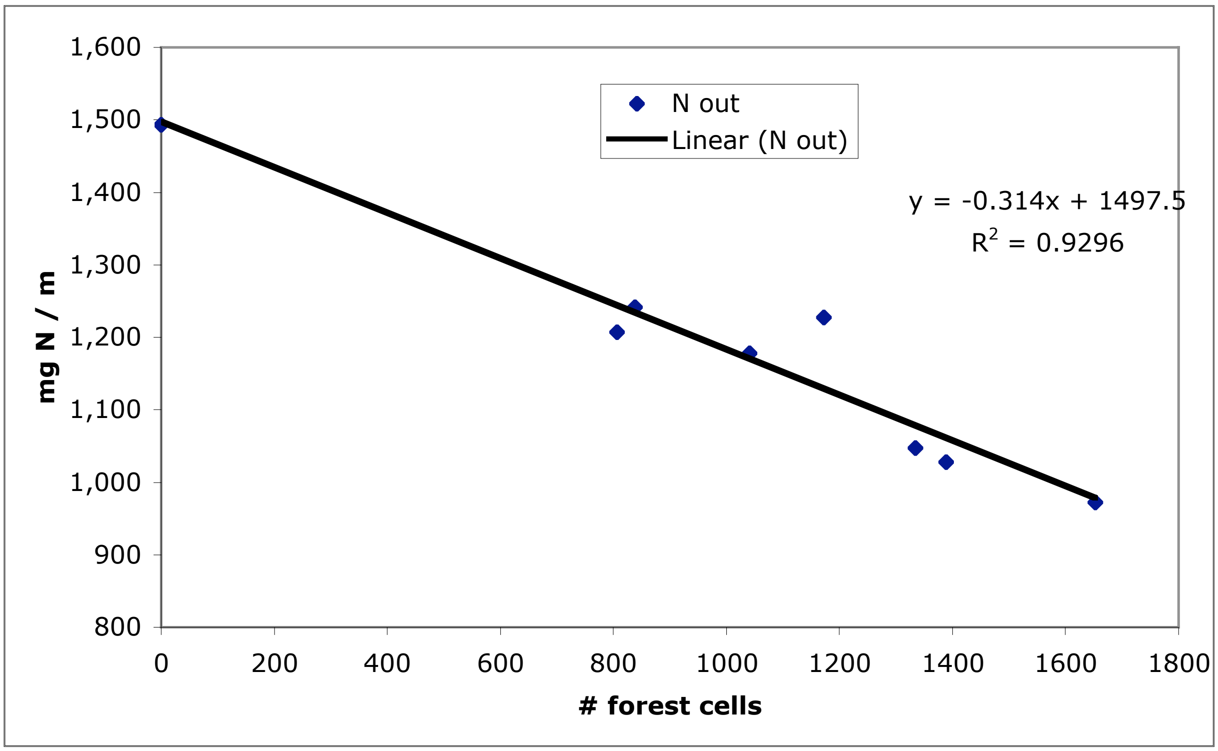
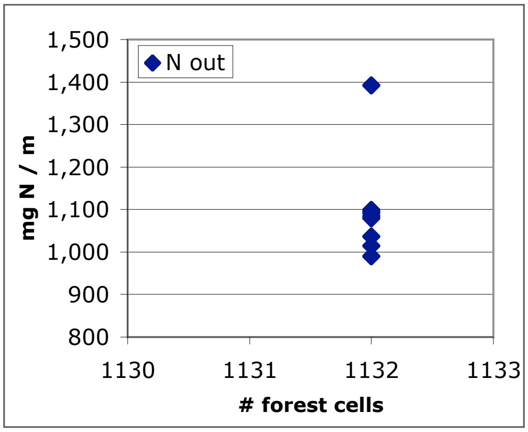
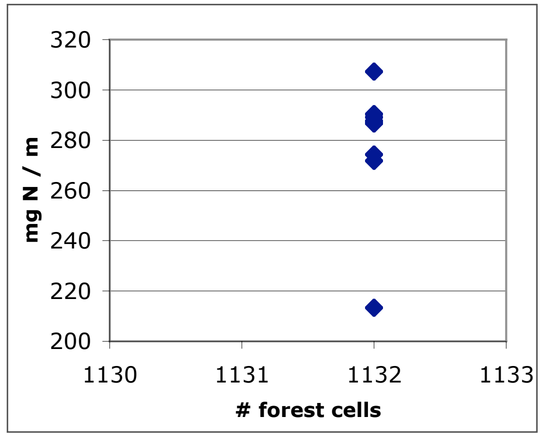


Figure 11: Response of total nitrogen in estuary to the number of forested cells on the watershed.





A. Estuary zone B. Gauging station at mid-watershed

Figure 12. Response of total nitrogen amounts to changes in pattern of forests in the watershed (Number of forested cells is 1,132 in both A and B.)

Although we did not create any scenarios that would specifically target the sub-watershed above the mid-watershed gauging station, we still get a response that is close to linear in terms of the total number of forested cells in the subwatershed (Figure 13). In this Figure we are looking at output from the same 17 scenarios described above, where the spatial variations were formulated for the whole watershed, making the subwatershed variations less clear in terms of buffer size and forest allocation. The deviations here are somewhat larger than in the previous case, when we were looking at the watershed as a whole (Figure 13). Local conditions tend to be more vulnerable to change than larger tracts of land.

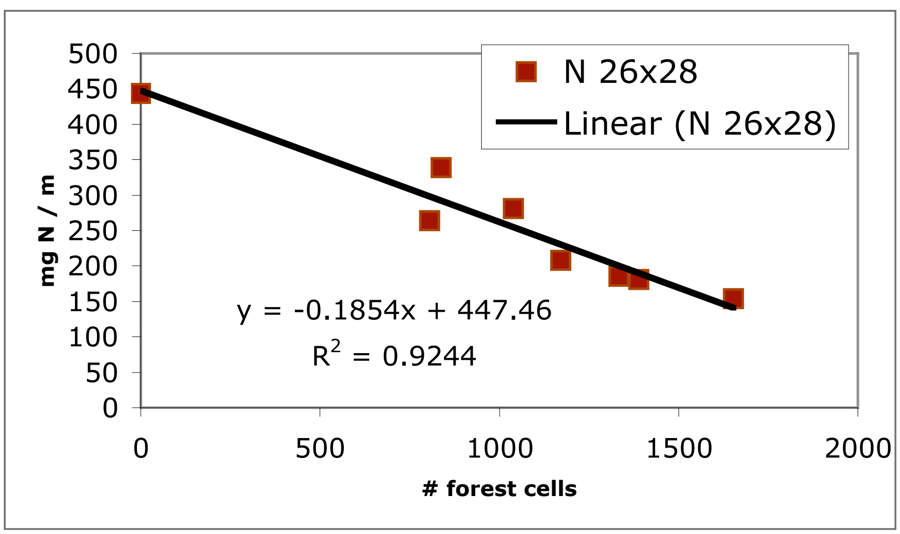


Figure 13: Relationship between water quality indicator at mid-watershed gauging station and overall land use patterns in the Hunting Creek watershed.

Conclusions

The results from these basic scenarios show, perhaps unsurprisingly, that the more forest is converted to agriculture, the poorer the quality of water in the estuary and the lower the NPP index. More generally, even this limited analysis shows that different land use allocations and patterns affect the level of ecosystem services generated in a given landscape; for the water quality index this difference can be as large as 40%.[12] Location is critical for some ecoservices; for example, forests located close to the estuary zone play a more important role in terms of estuary water quality than forests located far away. Ecosystem size is also important both by itself and as it interactions with location. For example, small river buffers have only a minor impact on water quality: the riparian buffers need to be large enough to be of use in maintaining water regulation services.

There is still a great deal of uncertainty in the estimated magnitudes of these effects. Different ecosystem services may be impacted differently by the same patterns and allocations. For example, while small forest patches appear to be better than large forest clusters from the viewpoint of water quality, a biodiversity index is very likely to favor larger patches. Much more detailed and comprehensive studies are required to take into account the whole range of ecosystem services and to account properly for all of the significant spatial heterogeneities and interactions.

The New Jersey landscape, like those in other places, is not homogeneous. Land uses appear in different patterns in different parts of the state, and these patterns may matter for specific ecosystem functions and the services that ecosystems produce. Spatial models such as the one described here can translate spatial land use allocations and patterns into indicators of the quantity and quality of ecosystem functions or services, enabling us to compare the impact of alternative landscape "design" or development patterns on overall ecosystem performance. Future studies could also include optimization experiments that would result in spatial land use allocations for New Jersey that would maximize the value of ecosystem services in defined geographic areas.

While the analyses reported here cannot yield quantitative ecosystem values for New Jersey, they clearly illustrate some of the ways in which factors external to an ecosystem can affect its functioning, and they also indicate one way in which those relationships can be modeled quantitatively. The analyses also highlight the importance of local land use planning and regulation in preserving and enhancing—or diminishing—the value of ecosystem services.

[12] This does not necessarily mean that the economic value associated with forest in the proximity of the estuary should be 40% higher than for upstream forest, since it is the specific combination of landscape conditions and spatial patterns that determines ecoservice levels.

Discussion

Natural Capital and Ecosystem Services

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). 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. In previous work (i.e. Costanza et al., 1989), we have displayed results using a range of discount rates and shown that a major source of uncertainty in the analysis is the choice of discount rate. But beyond this, there is 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 discoursing have also been proposed (i.e. Azar and Sterner, 1996, Sumaila and Walters, 2005, Weitzman 1998, Newell and Pizer 2003, 2004).

Table 16 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:

$$NPV = \sum_{t=0}^{\Sigma} V_t W_t \qquad (3)$$

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.

$$W_t = \left| \frac{1}{1+r} \right|^t \qquad (4)$$

Applying this formula to the annual ecosystem service flow estimates of \$10 Billion and \$15 Billion per year for a range of discount rates (r) from 0% to 8% yields the first two rows of estimates in Table 16. 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 17, 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 while an annual ecosystem service value of \$19 Billion for 100 years at a 0% discount rate yields an NPV of \$1.9 trillion. These estimates turn 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. As shown in Table 16, at an 8% discount rate an annual flow of \$11 billion translates to an NPV of \$138 billion and an annual flow of \$19 billion translates to an NPV of \$238 billion.

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, due to Weitzman (1998) and Newell and Pizer (2003, 2004) argues that discount rates are uncertain and because of this, their average value should be declining 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 due 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 in the future and its impact on welfare will be marginally less, due to the assumption of decreasing marginal returns to 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 to 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.

Table 16: 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
$19	$1,900	$1,900	$633	$380	$238
	Declining discount rate (300 yr time frame)				
$11		$1,809	$640	$299	$151
$19		$3,124	$1,106	$516	$261
	Intergenerational Discounting				
$11		$5,542	$870	$405	$212
$19		$9,572	$1,503	$699	$366

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. We do this by multiplying r by e^{-kt}, where k for this example was set to .00007. Since this function levels out at a discount rate of 0%, W_t eventually starts to increase again. We therefore forced W_t to level out at its minimum value. Also, carrying this calculation to infinity would also 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 inspection of Table 16, 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):

$$W_t = d^t + \frac{d * d^{t-1} * t}{G} \qquad (5)$$

Where:

$$d = \frac{1}{1+r} \tag{6}$$

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 NPV's are 5 times as much, while at 3% they are more than double.

There is no clear and unambiguous reason for choosing one of the three methods over the others, or for choosing a particular discount rate. 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. If we use 3% and focus on the two alternative methods, this would place the NPV of New Jersey's natural capital assets at somewhere between \$0.6 and \$1.5 trillion.

Reliability and Possible Sources of Error

Transferred value analysis estimates the economic value of a given ecosystem (e.g., wetlands) from prior studies of that ecosystem, most likely studies that were conducted in geographic areas other than the area being analyzed. Some have objected to this approach on the grounds that:

1. Every ecosystem is unique, and per-acre values derived from elsewhere in the world may not be relevant to the ecosystems being studied.
2. Even within a single ecosystem, the value per acre depends on the size of the ecosystem; in most cases, as the size decreases, the per-acre value would be expected to increase and vice versa. (In technical terms, 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). This issue was partly addressed in the spatial modelling component of this project, but this remains an important issue.
3. There is no way for us to obtain all of the data we would need to address these problems, and therefore we have no way of knowing the "true" value of all of the wetlands, forests, pastureland, etc. in a large geographic area and hence no way of knowing whether our estimated value is accurate or not and, if not, whether it is even high or low. In technical terms, we have far too few data points to construct a realistic demand curve or estimate a demand function.
4. To value all (or a large proportion) of the ecosystems in a large geographic area is questionable in terms of the standard definition of "exchange" value because we cannot conceive of a transaction in which all or most of a large area's ecosystems would be bought and sold. This emphasizes the point that the value estimates for large areas (as opposed to the unit values per acre) are more comparable to national income accounts aggregates and not exchange values (Howarth and Farber 2002). These aggregates (i.e. GDP) routinely impute values to public goods for which no conceivable market transaction is possible and it is just these kinds of aggregates that the value of ecosystem services of large geographic areas is comparable to (see below).

Unfortunately, the alternative recommended by those who advance the above arguments amounts to limiting valuation to a single ecosystem in a single location and using only data developed expressly for the unique ecosystem being studied, with no attempt to generalize to other 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.

The above objections to transferred value analysis have been responded to in detail elsewhere (Costanza et al 1998, Howarth and Farber 2002); the responses can be summarized as follows:

1. While every wetland, forest, etc. is obviously unique in some way, ecosystems of a given type

also by definition have many things in common. The use of average values in ecosystem valuation is no more and no less justified than their use in other "macroeconomic" contexts, e.g., developing economic statistics such as Gross State Product. This study's estimate of the aggregate value of New Jersey's ecosystem services is a valid and useful (albeit imperfect, as are all economic aggregates) basis for assessing and comparing these services with conventional economic goods and services.

2. The results of the spatial modeling analysis described later in this report do not support an across-the-board claim that the per-acre value of forest or agricultural land depends across-the-board 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, a major indicator of ecosystem health (and by implication of services tied to NPP), where each acre makes about the same contribution to the whole regardless of whether it is part of, e.g., a large forest patch or a small one. This area of inquiry certainly needs further research, but for the most part the assumption (that average value is a reasonable proxy for marginal value) seems appropriate as a first approximation.

3. As employed here, the prior studies we analyzed (most of which were peer-reviewed) encompass a wide variety of time periods, geographic areas, investigators, and analytic methods, and many of them provide a range of estimated values rather than single point estimates. 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. The approach is similar to defining an asking price for a piece of land based on the prices for "comparable" parcels; even though the property being sold is unique, realtors and lenders feel justified in following this procedure, even to the extent of publicizing a single asking price rather than a price range.

4. The objection as to the absence of even an imaginary exchange transaction was made in response to the study by Costanza et al. (1997) of the value of *all* of the world's ecosystems. Leaving that debate aside, one can in fact conceive of an exchange transaction in which all or a large portion of, e.g., New Jersey's wetlands was sold for development, so that the basic technical requirement that economic value reflect exchange value could in principle be satisfied. But even this is not necessary if one recognizes the different purpose of valuation at this scale – a purpose more analogous to national income accounting than to estimating exchange values (cf. Howarth and Farber 2002)

In the last analysis, this report takes the position that "the proof is in the pudding", i.e., the possibility of plausibly estimating the value of an entire state's ecosystem services is best demonstrated by presenting the results of an attempt to do so. In this report we have tried to display our results in a way that allows one to appreciate the range of values and their distribution (see, e.g., Tables 4 and 5). 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.[13]

In terms of more specific concerns, the value transfer methodology introduces an unknown level of error, because we usually do not know how well the original study site approximates conditions in New Jersey. Other potential sources of error in this type of analysis have been identified (Costanza et al.

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[13] The estimated value of the world's ecosystems presented in Costanza et al. (1997) has been criticized as both (1) "a serious underestimate of infinity" and (2) impossibly exceeding the entire Gross World Product. These objections seem difficult to reconcile.

1997) as follows:

1. Incomplete coverage is perhaps the most serious issue. Not all ecosystems have been well studied and some have not been studied at all as is evident from the gap analysis presented below. More complete coverage would almost certainly increase the values shown in this report, since no known valuation studies have reported estimated values of less than zero.
2. Distortions in current prices used to estimate ecoservice values are carried through the analysis. These prices do not reflect environmental externalities and are therefore again likely to be underestimates of "true" values.
3. 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.
4. The valuations probably underestimate shifts in the relevant demand curves as the sources of ecoservices become more limited. If New Jersey's ecosystem services are scarcer than assumed here, their value has been underestimated in this study. Such reductions in "supply" appear likely as land conversion and development proceed; climate change may also adversely affect New Jersey's ecosystems, although the precise impacts are harder to predict.
5. The valuations assume smooth responses to changes in ecosystem quantity with no thresholds or discontinuities. Assuming (as seems likely) that such gaps or jumps in the demand curve would move demand to higher levels than a smooth curve, the presence of thresholds or discontinuities would likely produce higher values for affected services (Limburg et al., 2002).
6. As noted above, the method used here assumes spatial homogeneity of services within ecosystems. The spatial modeling component of the project was intended to address this issue and showed that, indeed, the physical quantities of some services vary significantly with spatial patterns of land use and land cover. Whether this fact would increase or decrease valuations is unclear, and depends on the specific spatial patterns and services involved.
7. Our analysis uses a static, partial equilibrium framework that ignores interdependencies and dynamics. 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.
8. The value estimates are not necessarily based on sustainable use levels. Limiting use to sustainable levels would imply higher values for ecosystem services as the effective supply of such services is reduced.
9. The approach does not fully include the "infrastructure" or "existence" value of ecosystems. It is well known that people value the "existence" of certain ecosystems, even if they never plan to use or benefit from them in any direct way. But estimates of existence value are rare. Including this service would obviously increase the total values.
10. On a global level, there are great difficulties and imprecision in making inter-country comparisons. This problem was of limited relevance to the current project, since the majority of value transfer estimates were from the US or other developed countries.
11. In the few cases where we needed to convert from stock values to annual flow values, the amortization procedure also creates significant uncertainty, both as to the method chosen and the specific amortization rate used. (In this context, amortization is the converse of discounting.)
12. All of these valuation methods use static snapshots of ecosystems with no dynamic interactions.

The effect of this omission on valuations is difficult to assess.

13. Because the transferred value method is based on average rather than marginal cost, it cannot provide estimates consumer surplus. However, this means that valuations based on averages are more likely to underestimate total value.

If these problems and limitations could be addressed, the result would most likely be 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 estimate their value to be roughly twice that estimated by Costanza et al. (1997), which used a static, partial equilibrium analysis. Whether a similar result would obtain for New Jersey is impossible to say, but it does give an indication of the potential range of values.

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Appendix A: Literature Review

Ecosystem services are the benefits people obtain from ecosystems (Costanza et al. 1997, Daily 1997, de Groot et al. 2002). These include provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits (Millennium Ecosystem Assessment 2003).

Ecosystem services are becoming more scarce. 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 to 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. Since 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 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 (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.

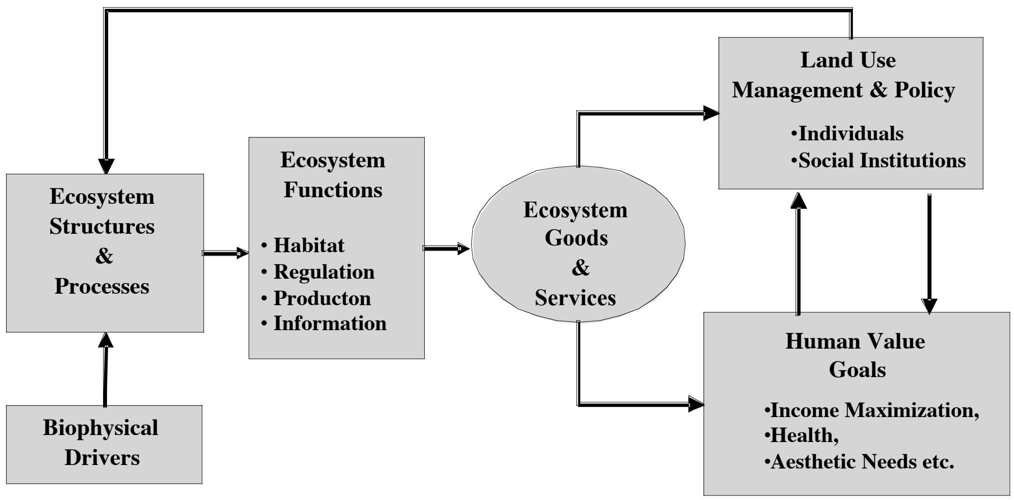

Figure 1: Framework for Integrated Assessment and Valuation of Ecosystem Goods and Services

Methodology for ESV

While measuring exchange values simply requires monitoring market data for observable trades, non-market values of goods and services are much more difficult to measure. Indeed, it is these values that have captured the attention of environmental and resource economists who have developed a number of techniques for valuing ecosystem goods and services (Freeman 2003, Bingham et al. 1995, Farber et. al. 2002, deGroot et al. 2002). 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 values when market values do not exist are identified in Table 1.

Table 1: Economic Valuation Techniques

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

Replacement Cost (RC): services could be replaced with man-made systems. For example, waste treatment can be replaced with costly treatment systems.

Net Factor Income (NFI): services provide for the enhancement of incomes; For example, water quality improvements may 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. For example, 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.

Contingent Valuation (CV): service demand may be elicited by posing hypothetical scenarios in surveys that involve some valuation of land use alternatives. For example, many people would be willing to pay for increased preservation of wildlife.

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

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 the descriptions in table 1 suggest, each valuation methodology has its own limitations, often limiting its use to a select range of ecosystem services. For example, the economic value generated by a naturally functioning ecological system can be estimated using the Replacement Cost (RC) method which is based on the price of the cheapest alternative way of obtaining that service, e.g., the value of a wetland in the treatment of wastewater might be estimated using the cost of chemical or mechanical alternatives. A related method, Avoided Cost (AC), can be used to estimate economic value based on the cost of damages due to lost services. 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. On the other hand, Contingent Valuation (CV) surveys are often employed in the absence of actual environmental use to estimate the economic value of less tangible services like critical wildlife habitat or amenity values. Marginal Product Estimation (MP) has generally been used in a dynamic modeling context and represents a helpful way to examine how ecosystem service values change over time. Finally, group valuation (GV) is a more recent addition to the valuation literature and directly addresses the need to measure social values directly in a group context. In many applications, the full suite of ecosystem valuation techniques will be required to account for the economic value of goods and services provided by a natural landscape.

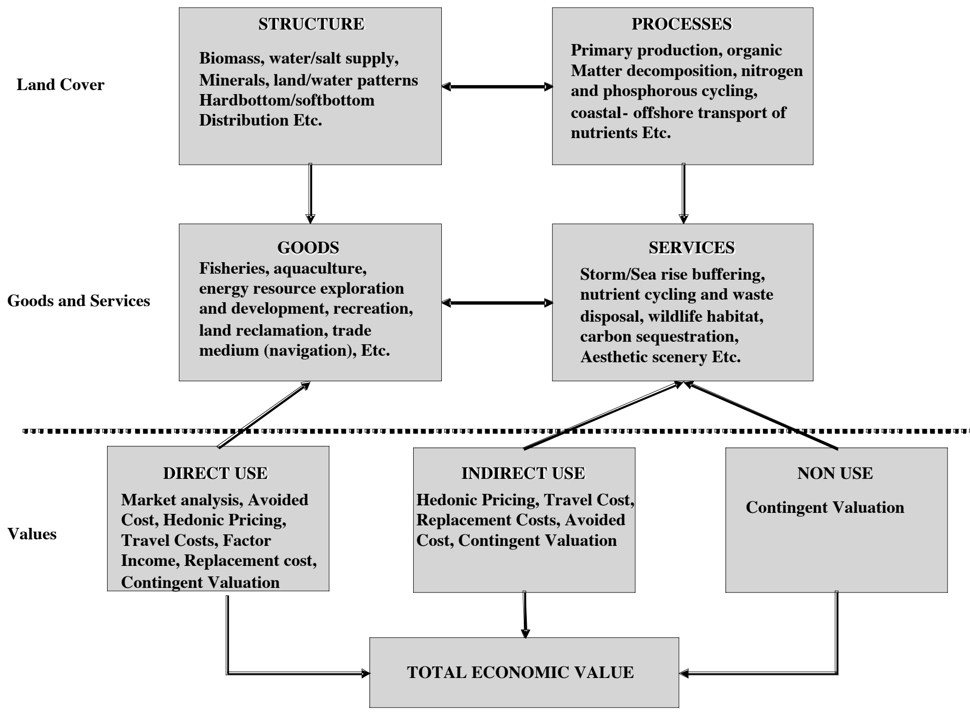

Figure 2: Total Economic Value of Ecosystem Functions, Goods and Services

Figure 2 depicts how the total value of a given landscape might be estimated by 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. Key linkages are made between the diverse structures and processes associated with the landscape and habitat features that created them and the goods and services that result. Once delineated, values for these goods and services can then be assessed by measuring the contribution they make to supporting human welfare. In economic terms, the natural assets of the landscape can thus yield direct (fishing) and indirect (nutrient regulation) use values as well as non-use (preservation) values of the system. Once accounted for, these economic values can then be aggregated to estimate the total value of the landscape (i.e. Total Economic Value or TEV as shown in Figure 2).

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[14]. Rather, we focus on how people faced the challenge presented by the transdisciplinary nature of

[14] Several reviews of the published ESV literature have been developed elsewhere. These review, including Smith

ESV research. In the 1960s, for instance, there was relatively little work that transcended disciplinary boundaries on ecosystem services. In later years this situation has gradually improved. Truly *transdisciplinary* approachs 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. see 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 3 and they are placed above the timeline, while below the line is a chronology of social events that may have triggered the development of ESV[15]. The chart is meant to be illustrative, not comprehensive, as space prohibits showing all important contributions and milestones.

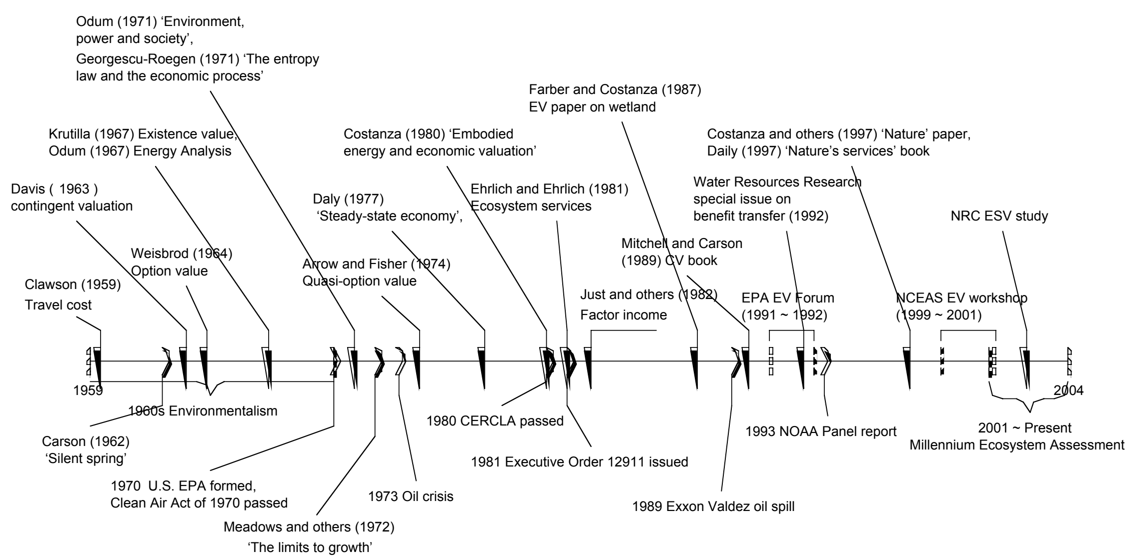

Figure 3. A historical overview of Ecosystem Valuation research

(1993, 2000), Carson (2000), Cropper (2000), Freeman (2003), Chapters in van den Bergh (1999), and Bateman and Willis (1999), provided much more detailed examination of ESV methods.

[15] In general we expect a time lag between the social events and relevant academic publications. For instance, Fisher and Ward (2000) assumed two years as the lag in the writing and publication process for their 'breakpoint analysis".

1960s—Common challenge, separate answers

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[16] at the time), economists began rethinking the role of 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 being considered. These include option value, 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).

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 completely 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

[16] 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 the number of individuals form a species increases dramatically.

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. Economists of that 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 Lenard 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 *Factor Income* (or the Productivity-based method) 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 marketed good.

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[17].

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 of 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 4, the 1980s witnessed the dramatic increases in the number of publications, including peer-reviewed papers, book chapters, governmental reports and thesis, on the topic of ecosystem valuation[18]. This result is based on a search in the Environmental Valuation Reference

[17] Recent progress in the area includes Barbier (1994) , Barbier and Strand (1998), Barbier (2000), Knowler et al. (2003),

[18] The drop of the total number of publications since late 1990s is probably due to artificial effect, i.e. EVRI™ has not incorporated all the papers in recent years. According to a similar analysis by Adamowicz (2004), the number of peer-reviewed literature in environmental valuation has increased over time and did not decrease after 1995. In addition, the same paper showed the growth in valuation publication is not solely the result of a larger number of

Inventory TM (EVRITM), 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 built environment from EVRITM because they are not relevant to ESV.

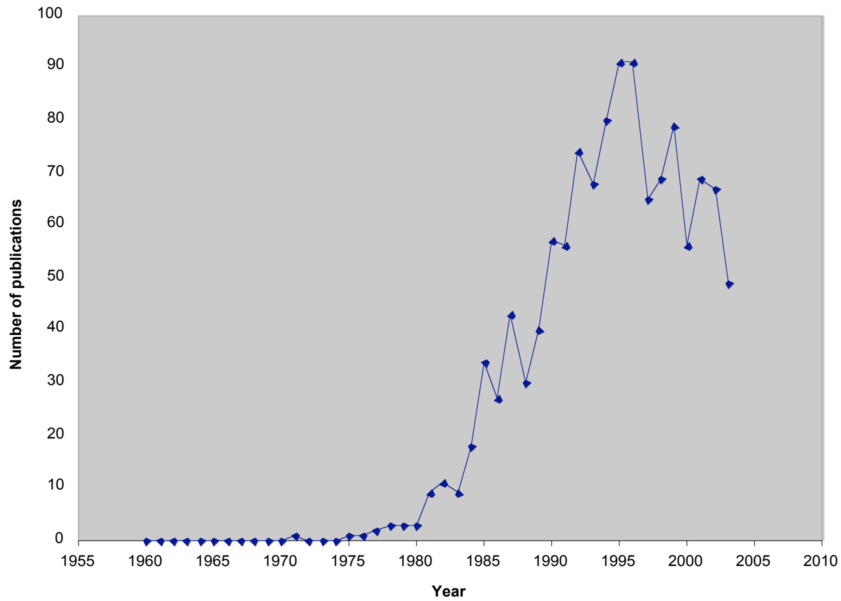

Figure 4. Number of ESV publications in EVRI over time.

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 changed rather abruptly (Carson et al. 2003). In the same year, two leading researchers published their start-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 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[19]) 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

total publications.

[19] 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.

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" as a key word (in both singular and plural forms), we did a search in the ISI Web of Knowledge. Figure 5 shows the total number of papers published and the number of disciplinary categories in which they occur over time. For example, the curves show that by the year 2003, close to 70 papers per year were being published on ecosystem services - in more than 40 subdisciplines[20]. 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.

The concept of ecosystem services (and the related concept of "natural capital[21]" which first appeared in Costanza and Daly (1982)) have proven useful for landscape management and decision making for two fundamental reasons. First, they help synthesize essential ecological and economic concepts, allowing researchers and managers to link human and ecological systems in a viable and policy relevant manner. Second, scientists and policy makers can use the concepts to evaluate economic and political tradeoffs between landscape development and conservation alternatives.

1990s ~ present: Moving toward trandisciplinary ESV research

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 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).

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[20] This number is almost for sure an under-estimate because similar terms such as "ecological service(s)" and environmental service(s)' were not included.

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

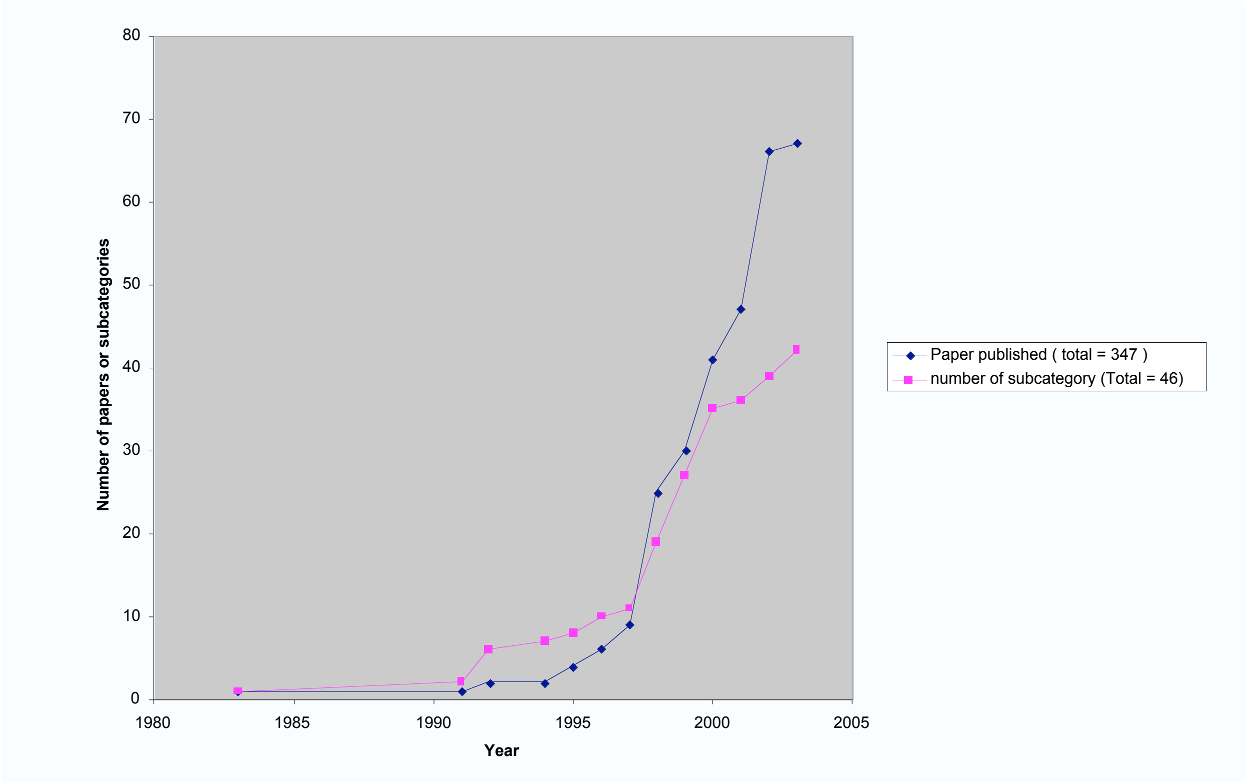

Figure 5. Number of peer-reviewed ecosystem service papers and their related sub-categories over time listed in the ISI Web of Science.

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 gained its 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 characteristics has a monetary price, then it is possible to compute the respondent's willingness to pay for other attribute.

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 being 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 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 measure individual preferences but from a process of open public debate (Jacobs 1997, Coote and Lenaghan 1997). This method is extremely useful in ESV addressing 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).

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). They 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. The method is well-suited to both eliciting values and preferences and evaluating stakeholder interests.

Traditional MCDA assumes that there is a single decision-maker so that clear, unambiguous, non-conflicting objectives can be identified from a single perspective. Furthermore, it is assumed that the relevant criteria are well-defined, independent of each other, and measurable with certainty (Stewart 1995). In order to extend MCDA to group decision situations where there are conflicting objectives and to incorporate uncertainty into the decision-making process, MCDA needs to be used in conjunction with discursive participatory methods and with ecosystem modeling.

Fernandes et al. (1999) provide an example of MCDA in a participatory setting for coral reef management in Saba Marine Park, an island in the Netherlands Antilles. The process provided a forum for tabling, discussing and documenting the community's concerns and allowed the unexpected degree of general agreement to become apparent. In this sense, it facilitated social discourse, value formation and learning about the interactions of the social, economic and ecological systems.

The emergence of these new interdisciplinary methods can be attributed in part to two workshops in 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 field of ecology, economics, and philosophy offered guidelines for ESV, among which "Economists and ecologists should work together from the very beginning to ensure the output from ecological model is in a form that can be used as input for economic model" (Water Science and Technology Board 2004). 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 on valuing the services provided by global ecosystems in *Nature*. They estimated that the annual value of 17 ecosystem services for the entire biosphere was US$33 trillion (Costanza et al. 1997). The journal of *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* brings together world-renowned scientists from a variety of disciplines to examine the character and value of ecosystem services, the damage that has been done to them, 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.

State-of-the-art ESV- Millennium Ecosystem Assessment

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. Instead of evaluating how ecosystems respond to just one environmental concern, such as climate change, the experts will attempt to provide a complete "planetary health check", that identifies and where possible quantifies the impacts of changes in land use, loss of biodiversity, the application of agricultural fertilizers, and many other factors. The synthesis report now is available for review at http://www.millennium assessment.org/en/index.aspx.

ESV in Practice

In the ESV area most of the final demand comes from policy makers and public agencies[22]. To what extent, however, is ESV actually used to make real environmental decisions?

The answer to this question is contingent on the specific area of environmental policy making that is of concern. There are a few areas in which ESV is well established. They 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 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.

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).

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

[22] 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), and Adamowicz (2004).

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). Their results show that the use of CBA has increased substantially in the last decade. Ten years ago, one project in 162 used CBA. In comparison, as many as one third of the projects in the environmental portfolio did so in recent years[23]. While this represents a substantial improvement, the authors predicted "there remains considerable scope for growth" (p1).

At the Macro-economic level, ESV has been used in setting up carbon taxes and calculating 'Green" GDP[24] (Pearce 1993). For the purpose of this paper, we will focus on ESV's micro-level roles in (1) Natural Resource Damage Assessments (NRDA), (2) CBA/CEA (Cost Effectiveness Analysis), (3) value transfer, and (4) GIS and ecosystem modeling. Since there are no specific mechanisms that track when research is used for policy, we have to rely on examples.

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 (see 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 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.

Although statutory authorities existed prior to the 1989 *Exxon Valdez* oil spill, the spill was a signal event in the development of trustee NRDA programs. In the years following the spill, NRDA has been on the frontier 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 (see 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).

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[23] 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).

[24] Demands that the accounts measure a green GDP reflect a desire to include more of the final non-market services in measures of national income. At the mean time, measure a green GDP could also mean including damage and/or degradation of ecosystem services (CBO 1994)

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)[25].

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 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 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 interactions 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.

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[25] 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).

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 source for all CBAs of that agency's regulations 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 rule makings 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 value transfer

V*alue transfer* (or benefit transfer) is defined as the adaptation of existing ESV information or data to new policy contexts that 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, Desvouges 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 value transfer, as a "second-best" strategy, is important to evaluating management and policy impacts. For instance, EPA's regulation development process almost always involves value transfer. Although it is explicitly recognized in the EPA's *Guidelines for Preparing Economic Analyses (2000)* that this is not the optimal situation, but conducting an original study for anything but the largest regulation 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).

However, many original valuation studies are not designed for application purpose in the comparative framework that is inherent to the value transfer method, making the identification and recovery of suitable empirical studies for transfer difficult. In fact, in many cases valuation estimates are generated as a by-product of efforts to clarify research methods (McConnell 1992). This has resulted in a somewhat paradoxical situation in the peer-reviewed economic valuation literature that when a methodology is well understood and achieves reasonably high levels of professional acceptance, the attention of editors and readers shifts to new issues. As a result, peer-reviewed publications often serve merely as a vehicle for illustrating the most recent valuation method. Little interest is expressed in replication of studies or in new applications of previously developed methods, the very things which are required for developing policy for sites and actions not explicitly involved in the original study (Smith 1992).

This problem could be partly solved by constructing databases that collect ESV information for the purpose of value transfer[26]. Recognizing the widespread need for a non-market valuation library, Environment Canada, in collaboration with the U.S. Environmental Protection Agency and leading North American experts, has developed a value transfer database: the Environmental Valuation Reference Inventory ™ (EVRI™) {De Civita, 1998 #92}. Other similar efforts include the EnValue database sponsored by New South Wales Environmental Protection Authority in Australia (http://www.epa.nsw.gov.au/envalue/) and the ocean-related ESV database of National Ocean Economic Program (http://essp.csumb.edu/noep/index.html). As acknowledged by these websites, care must be taken in transferring database values to other sites, and there is neither a generally accepted verdict on the utility of these efforts to date or on a value transfer protocol in general.

Integration with GIS and Modeling

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). In doing so, the value transfer process is augmented with 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.

Thanks to the increased ease of using Geographic Information Systems (GIS) and the public availability of land cover data sets derived from satellite images, geographic information can more easily be attributed with ecosystem service values. 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 (Wilson et al. 2003) 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

[26] In addition, development of more transferable value measures and further development of value transfer techniques is also very important.

are then used to estimate the economic value of ecosystem services (see 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 working in 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 (Massachusetts Audubon Society 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 recently 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 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.

Debate on the use of ESV

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

1. to provide for comparisons of contributions of natural capital to human welfare with those of physical and human capital.
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 in terms 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 proceed 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). As a one-dimensional concept, based exclusively on individual's preferences, the principle of maximizing expected utility is judged to be inadequate and 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 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 of 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 since, 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.

Their usefulness lies in the fact that they use 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, science).

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 multiple attribute decision making (MADM) 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 with have to be evaluated one way or the other. A MADM 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.

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[27].

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 term and in 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, science) argued, it should be considered as a framework and a set of procedures to help organize available information. Viewed in this light, benefit-cost analysis does not dictate choices; nor does it replace the ultimate authority and responsibility of decisonmakers. It is simply a tool for organizing and expressing certain kinds of information on the 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 decisionmakers 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 (Freeman 2003)

Methodological and technical debate

ESV has been also been criticized on methodological and technical grounds. There are a range of issues here which are covered in detail elsewhere (cf. Costanza et al. 1997b, Costanza 1998, Costanza et al. 1998, Pearce 1998, Bockstael et al. 2000, Costanza and Farber 2002). 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) contend 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 possible 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).

[27] 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).

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

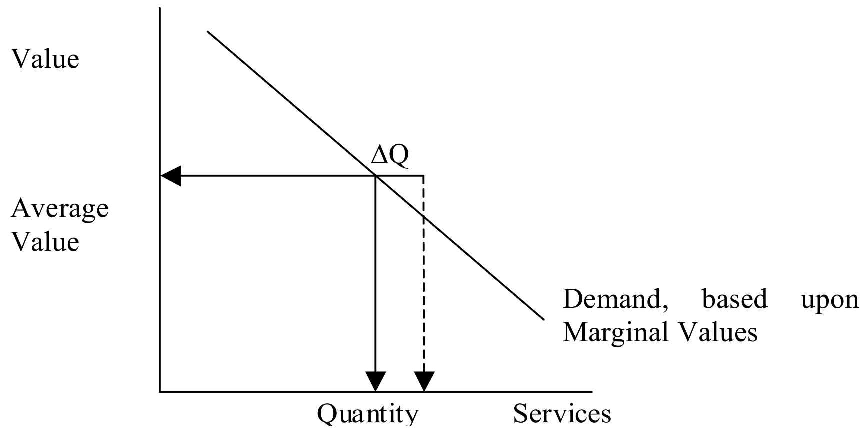

Figure 6. A model of Ecosystem Service Valuation

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" to human welfare of the service, since 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 one's 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 economic markets would likely feedback on the Demand for the 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. The "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 impacts 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-estimate depends again on the replaceability of the service being gained or lost.

Beyond the purpose confusion, 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 7 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.

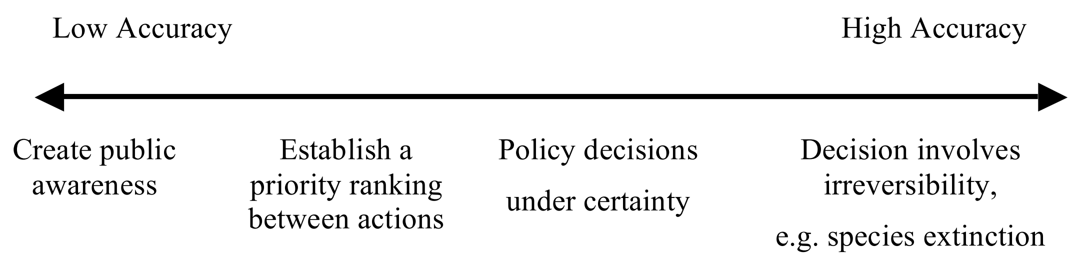

Figure 7. Accuracy Continuum for the ESV (adapted from Desvousges and Johnson 1998)

For example, using ESV to assist an environmental policy decision-maker in setting broad priorities for assessment and possibly 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 represent a gain in knowledge that costs society relatively little if the ESV results are later found to be inaccurate. As Figure 6 also shows, 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 in research—the need for a transdisciplinary approach

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 accept that disciplinary boundaries are academic constructs that are irrelevant outside of the university and 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 (Boumans et al. 2002 is one example). But it is just this type of study that is of greatest relevance to decision makers (Turner et al. 2003) and looks to be the way forward.

ESV in practice—moving beyond the efficiency goal

We attempted to quantify ESV's contribution to environmental policy-making by answering questions like "to what extent is ESV actually used to make real decisions?" However, we soon realized that this goal was too ambitious. Instead, along with other reviewers (e.g. Pearce and Seccombe-Hett 2000, Adamowicz 2004), we 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 micro-level studies, including NRDA, CBA-CEA, value transfer analysis, and studies integrating ESV with GIS and/or ecosystem modeling.

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), and the practice of integrating ESV with GIS and ecosystem modeling (Bockstael et al. 1995, Costanza et al. 2002, Boumans et al. 2002, Wilson et al. 2004).

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Appendix C. Value Transfer Detailed Reports

New Jersey Value-Transfer Detailed Report (Type A)

			2004 dollars per acre/year				
Land Cover	**Author(s)**	**Method**	**Min**	**Max**	**Single Value**	**Mean**	**Median**
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**
	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

Land Cover	Author(s)	Method	Min	Max	Single Value	Mean	Median
			2004 dollars per acre/year				
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., Grigalunas, T. A. and Mazzotta, M. J.	CV			$1,462	$1,462	$1,462
					Aesthetic & Recreational	**$303**	**$158**
					Estuary Total	**$715**	**$281**

Land Cover		Author(s)	Method	2004 dollars per acre/year Min	Max	Single Value	Mean	Median
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
		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

Land Cover	Author(s)	Method	Min	Max	Single Value	Mean	Median
			2004 dollars per acre/year				
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**
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

Land Cover	Author(s)	Method	Min	Max	Single Value	Mean	Median
					2004 dollars per acre/year		
	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
	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

Land Cover	Author(s)	Method	Min	Max	Single Value	Mean	Median
			2004 dollars per acre/year				
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**
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**

Land Cover		Author(s)	Method	Min	Max	2004 dollars per acre/year Single Value	Mean	Median
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
		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**

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
						2004 dollars per acre/year		
Open freshwater, cont.	*Aesthetic & Recreational*							
		Young, C. E. and Shortle, J. S.	HP			$70	$70	$70
		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**

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Pasture								
	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
		Rein, F. A.-1999	AC	$36	$158		$97	$97

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Riparian buffer, cont.		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
		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**

Land Cover		Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
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
						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

Land Cover	Author(s)	Method	Min	Max	Single Value	Mean	Median
			2004 dollars per acre/year				
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**
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**

Land Cover		Author(s)	Method	Min	Max	2004 dollars per acre/year Single Value	Mean	Median
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
MD	Multiattribute Decision Analysis
EA	Energy Analysis
MP	Marginal Product Estimation
CRS	Combined Revealed and Stated Preference
MA	Meta-anlaysis
VT	Value Transfer

New Jersey Value-Transfer Detailed Report (Type A-C)

Land Cover	Ecosystem Service	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	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	TC	$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**	**$38,003**
Coastal Shelf								
	Water supply							
		Soderqvist, T. and Scharin, H.	CV	$243	$404		$323	$323
		Nunes, P and Van den Bergh, J.	CV			$517	$517	$517

				2004 dollars per acre/year				
Land Cover	**Ecosystem Service**	**Author(s)**	**Method**	**Min**	**Max**	**Single Value**	**Mean**	**Median**
Coastal shelf, cont.		Hanley, N., Bell, D. and Alvarez-Farizo, B.	CRS			$723	$724	$724
						Water supply	**$521**	**$517**
	Biological Control	Costanza, R. et. al.-1997	VT			$20	$20	$20
						Biological control	**$20**	**$20**
	Nutrient regulation	Costanza, R. et. al.-1997	VT			$723	$723	$723
						Nutrient regulation	**$723**	**$723**
	Cultural & Spiritual	Costanza, R. et. al.-1997	VT			$35	$35	$35
						Gas & Climate regulation	**$35**	**$35**
						Coastal Shelf Total	**$1,299**	**$1,295**
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**
	Biological Control	Costanza, R. et al.	VT	$12	$12		$12	$12
						Biological control	**$12**	**$12**
	Refugium function & Wildlife conservation							
		Christie, M., Hanley, N, Warren, J., et al.	CV			$1,242	$1,242	$1,242

Land Cover	Ecosystem Service	Author(s)	Method	2004 dollars per acre/year Min	Max	Single Value	Mean	Median
Cropland, cont.		Christie, M., Hanley, N, Warren, J., et al.	CV			$419	$419	$419
						Refugium function	**$831**	**$831**
	Aesthetic & Recreational	Bergstrom, J., Dillman, B. L. and Stoll, J. R.-1985	CV			$26	$26	$26
		Alvarez-Farizo, B., Hanley, N., Wright, R. E. et al.	CV			$4	$4	$4
						Aesthetic & Recreational	**$15**	**$15**
						Cropland Total	**$866**	**$865**
Estuary								
	Disturbance prevention	Costanza, R. et. al.-1997	VT	$286	$286		$286	$286
						Disturbance prevention	**$286**	**$286**
	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**
	Nutrient regulation	Costanza, R. et. al.-1997	VT	$10,658	$10,658		$10,658	$10,658
						Nutrient regulation	**$10,658**	**$10,658**

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Estuary, cont.	*Biological Control*	Costanza, R. et. al.-1997	VT	$39	$39		$39	
						Biological Control	**$39**	**$39**
	Refugium function	Johnston, R. J. et. al.-2002	MP			$1,298	$1,298	$1,298
		Johnston, R. J. et. al.-2002	MP			$82	$82	$82
		Johnston, R. J. et. al.-2002	MP			$412	$412	$412
		Farber, S. and Costanza, R.-1987	MP			$11	$11	$11
		Farber, S. and Costanza, R.-1987	MP			$15	$15	$15
		Costanza, R. et. al.-1997	VT	$66	$66		$66	$66
						Refugium function	**$314**	**$74**
	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			$158	$158	$158
		Johnston, R. J. et. al.-2002	TC			$219	$219	$219
		Johnston, R. J. et. al.-2002	TC			$333	$333	$333

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	2004 dollars per acre/year Single Value	Mean	Median
Estuary, cont.		Johnston, R. J. et. al.-2002	CV			$1,462	$1,462	$1,462
	.	Costanza, R. et. al.-1997	VT	$192	$192		$192	$192
				Aesthetic & Recreational			**$292**	**$175**
	Cultural & Spiritual	Costanza, R. et. al.-1997	VT	$15	$15		$15	$15
				Cultural & Spiritual			**$15**	**$15**
				Estuary Total			**$11,653**	**$11,289**
Forest								
	Gas & Climate regulation	Reyes, J. and Mates, W.-2004	VT			$11	$11	$11
		Pimentel, D.-1998	AC			$13	$13	$13
		Tol, R. S. J.	MP			$57	$57	$57
		Tol, R. S. J.	MP			$302	$302	$302
		Tol, R. S. J. and Downing, T. E.	MP			$26	$26	$26
		Tol, R. S. J. and Downing, T. E.	MP			$16	$16	$16
		Tol, R. S. J. and Downing, T. E.	MP			$74	$74	$74
		Tol, R. S. J. and Downing, T. E.	MP			$20	$20	$20
		Tol, R. S. J. and Downing, T. E.	MP			$78	$78	$78
		Tol, R. S. J. and Downing, T. E.	MP			$1	$1	$1
		Schauer, M. J.	MP			$23	$23	$23
		Schauer, M. J.	MP			$318	$318	$318
		Roughgarden, T. & Schneider, S.	MP		$184	$39	$39	$39

				2004 dollars per acre/year				
Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
Forest, cont.		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.	MP	$10	$46	$20	$20	$20
		Plambeck, E. L.	MP	$371	$933	$419	$419	$419
		Nordhaus, W. D. and Popp, D.	MP	$0.04	$32	$11	$11	$11
		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	$8	$66	$33	$33	$33
		Nordhaus, W. D.	MP	$2	$15	$7	$7	$7
		Nordhaus, W. D.	MP	$0.31	$2	$1	$1	$1
		Newell, R. G. and Pizer, W. A.	MP	$10	$34		$22	$22
		Newell, R. G. and Pizer, W. A.	MP	$7	$23		$15	$15
		Maddison, D.	MP			$16	$16	$16
		Hope, C. and Maul, P.	MP	$11	$43	$28	$28	$28
		Fankhauser, S.	MP	$6	$43	$19	$19	$19
		Fankhauser, S.	MP	$23	$66	$40	$40	$40

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Forest, cont.		Fankhauser, S.	MP	$5	$37	$17	$17	$17
		Costanza, R., et al.	MP	$44	$44		$44	$44
		Azar, C. and Sterner, T.	MP			$202	$202	$202
		Azar, C. and Sterner, T.	MP			$10	$10	$10
		Azar, C. and Sterner, T.	MP			$30	$30	$30
		Azar, C. and Sterner, T.	MP			$66	$66	$66
				Gas & Climate regulation			**$12**	**$12**
	Soil formation	Costanza, R. et. al.-1997	VT	$5	$5		$5	$5
						Soil formation	**$5**	**$5**
	Waste treatment	Costanza, R. et. al.-1997	VT	$44	$44		$44	$44
						Waste treatment	**$44**	**$44**
	Pollination	Hougner, C.	RC	$59	$265		$162	$162
						Pollination	**$162**	**$162**
	Biological Control	Costanza, R. et. al.-1997	VT	$2	$2		$2	$2
						Biological Control	**$2**	**$2**
	Refugium function	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

Land Cover	Ecosystem Service	Author(s)	Method	2004 dollars per acre/year Min	Max	Single Value	Mean	Median
Forest, cont.		Garrod, G., D. and Willis K. G.	CV	$3,101	$3,383		$3,242	$3,242
		Garrod, G. D. and Willis. K. G.	CV			$15	$15	$15
		Garrod, G. D. and Willis K. G.	CV	$1,817	$2,003		$1,910	$1,910
		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
					Refugium function		**$923**	**$279**
	Aesthetic & Recreational	Willis, K. G.-1991	TC	$0	$1		$1	$1
		Willis, K. G.-1991	TC	$20	$35		$28	$28
		Willis, K. G.-1991	TC	$8	$15		$12	$12
		Willis, K. G.-1991	TC	$5	$5		$5	$5
		Willis, K. G.-1991	TC	$89	$162		$126	$126
		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
		Costanza, R. et. al.-1997	VT	$18	$18		$18	$18
		Boxall, P. C., McFarlane, B. L. and Gartrell, M.-1996	TC			$0	$0	$0

				2004 dollars per acre/year				
Land Cover	**Ecosystem Service**	**Author(s)**	**Method**	**Min**	**Max**	**Single Value**	**Mean**	**Median**
Forest, cont.		Bishop, K.-1992	CV			$543	$543	$543
		Bishop, K.-1992	CV			$485	$485	$485
	.	Bennett, R., et. al.-1995	CV			$144	$144	$144
						Aesthetic & Recreational	**$122**	**$12**
	Cultural & Spiritual	Costanza, R. et. al.-1997	VT	$1	$1		$1	$1
						Cultural & Spiritual	**$1**	**$1**
	Water supply	NJEPA et al.-2005	RC	$247	$385		$316	$316
		Loomis, J. B.	TC	$9	$9		$9	$9
						Water Supply	**$163**	**$163**
						Forest Total	**$1,476**	**$688**
Freshwater Wetland								
	Gas & Climate regulation	Costanza, R. et. al.-1997	VT	$134	$134		$134	$134
						Gas & Climate regulation	**$134**	**$134**
	Disturbance prevention	Costanza, R. et. al.-1997	VT	$3,657	$3,657		$3,657	$3,657
						Disturbance prevention	**$3,657**	**$3,657**
	Water regulation	Thibodeau, F. R. and Ostro, B. D.-1981	AC			$5,957	$5,957	$5,957
		Costanza, R. et. al.-1997	VT	$15	$15		$15	$15

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
							2004 dollars per acre/year	
Freshwater wetland, cont						***Water regulation***	**$2,986**	**$2,986**
	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
		Costanza, R. et. al.-1997	VT	$3,839	$3,839		$3,839	$3,839
						Water supply	**$1,544**	**$1,401**
	Waste treatment	Costanza, R. et. al.-1997	VT	$838	$838		$838	$838
						Waste treatment	**$838**	**$838**
	Refugium function	Vankooten, G. C. and Schmitz, A.-1992	CV			$5	$5	$5
		Costanza, R. et. al.-1997	VT	$222	$222		$222	$222
						Refugium function	**$113**	**$113**
	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

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Freshwater wetland, cont.		Thibodeau, F. R. and Ostro, B. D.-1981	TC	$27	$86		$56	$56
		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
		Costanza, R. et. al.-1997	VT	$248	$248		$248	$248
						Aesthetic & Recreational	**$1,406**	**$950**
	Cultural & Spiritual	Costanza, R. et. al.-1997	VT	$890	$890		$890	$890
						Cultural & Spiritual	**$890**	**$890**
						Freshwater Wetland Total	**$11,568**	**$10,969**
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.-1988	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**

Land Cover	Ecosystem Service	Author(s)	Method	2004 dollars per acre/year Min	Max	Single Value	Mean	Median
Open freshwater, cont.	*Aesthetic & Recreational*							
		Young, C. E. and Shortle, J. S.-1989	HP			$70	$70	$70
		Ward, F. A., Roach, B. A. and Henderson, J. E.-1996	TC	$17	$1,635		$826	$826
		Shafer, E. L. et. al. -1993	TC			$938	$938	$938
		Shafer, E. L. et. al. -1993	TC			$470	$470	$470
		Shafer, E. L. et. al. -1993	CV			$83	$83	$83
		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	$115	$242		$179	$179
		Cordell, H. K. and Bergstrom, J. C.-1993	CV	$162	$679		$420	$420
		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**

Land Cover	Ecosystem Service	Author(s)	Method	2004 dollars per acre/year Min	Max	Single Value	Mean	Median
Open freshwater, cont.				**Open Fresh Water Total**			**$765**	**$781**
Pasture								
	Gas & Climate regulation	Costanza, R. et. al.-1997	VT	$4	$4		$4	$4
		Costanza, R. et. al.-1997	VT	$0	$0		$0 $0	
		Sala, O. E. and Paruelo, F. M.	MP	$4	$10	$5	$5	$5
				Gas & Climate regulation			**$3**	**$4**
	Water regulation	Costanza, R. et. al.-1997	VT	$2	$2		$2	$2
				Water regulation			**$2**	**$2**
	Soil formation	Pimentel, D.-1998	DM			$6	$6	$6
		Costanza, R. et. al.-1997	VT	$1	$1		$1	
				Soil formation			**$3**	**$3**
	Waste treatment	Costanza, R. et. al.-1997	VT	$44	$44		$44	$44
				Waste treatment			**$44**	**$44**
	Pollination	Costanza, R. et. al.-1997	VT	$13	$13		$13	$13
				Pollination			**$13**	**$13**
	Biological Control	Costanza, R. et. al.-1997	VT	$12	$12		$12	$12

Land Cover	Ecosystem Service	Author(s)	Method	2004 dollars per acre/year Min	Max	Single Value	Mean	Median
Pasture, cont.						***Biological Control***	**$12**	**$12**
	Aesthetic & Recreational	Costanza, R. et. al.-1997	VT			$1	$1	$1
		Boxall, P. C.-1995	TC			$0.03	$0.03	$0.03
		Alvarez-Farizo, B., Hanley, N., Wright, R. E. et al.				$1	$1	$1
						Aesthetic & Recreational	**$1**	**$1**
						Pasture Total	**$77**	**$77**
Riparian Buffer								
	Disturbance prevention	Rein, F. A.-1999	AC	$6	$99		$53	$53
		Rein, F. A.-1999	AC	$45	$201		$123	$123
						Disturbance prevention	**$88**	**$88**
	Water supply	Rich, P. R. and Moffitt, L. J.-1982	HP			$4	$4	$4
		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
		Gramlich, F. W.-1977	CV			$188	$188	$188
		Danielson, L., et. al.-1995	CV			$4,095	$4,095	$4,095

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Riparian buffer, cont.		Berrens, R. P., Ganderton, P. and Silva, C. L.-1996	CV			$1,794	$1,794	$1,794
		Kahn, J. R. and Buerger, R. B.-1994	TC	$3	$9		$6	$6
		Kahn, J. R. and Buerger, R. B.-1994	TC	$0.15	$1		$0.46	$0.46
						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
		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	$6,409
						Aesthetic & Recreational	**$1,370**	**$608**
	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**

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Saltwater Wetland								
	Disturbance prevention	Farber, S.-1987	AC	$1	$1		$1	$1
		Farber, S. and Costanza, R.-1987	AC			$1	$1	$1
		Costanza, R. et. al.-1997	VT	$929	$929		$929	$1
						Disturbance prevention	**$310**	**$1**
	Waste treatment	Costanza, R. et. al.-1997	VT	$3,382	$3,382		$3,382	$3,382
		Breaux, A., Farber, S. and Day, J.-1995	AC			$16,560	$16,560	$16,560
		Breaux, A., Farber, S. and Day, J.-1995	AC	$103	$116		$109	$109
		Breaux, A., Farber, S. and Day, J.-1995	AC	$1,256	$1,942		$1,599	$2,491
						Waste treatment	**$5,413**	**$2,491**
	Refugium function	Lynne, G. D., Conroy, P. and Prochaska, F. J.-1981	ME			$1	$1	$1
		Farber, S. and Costanza, R.-1987	ME			$1	$1	$1
		Costanza, R. et. al.-1997	VT	$85	$85		$85	$85
		Bell, F. W.-1997	FI	$144	$953		$549	$549
		Batie, S. S. and Wilson, J. R.-1978	ME	$6	$735		$370	$370
						Refugium function	**$201**	**$85**

Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
				2004 dollars per acre/year				
Saltwater wetland, cont.	*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			**$6,131**	**$2,771**
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			$164	$164	$164
		McPherson, E. G.-1992	AC			$820	$820	$820
				Gas & Climate regulation			**$336**	**$164**
	Water regulation	McPherson, E. G.-1992	AC			$6	$6	6
				Water regulation			**$6**	**$6**
	Aesthetic & Recreational	Tyrvainen, L.-2001	CV			$3,465	$3,465	$3,465
		Tyrvainen, L.-2001	CV			$1,182	$1,182	$1,182

						2004 dollars per acre/year		
Land Cover	Ecosystem Service	Author(s)	Method	Min	Max	Single Value	Mean	Median
Urban Green Space, cont.		Tyrvainen, L.-2001	CV			$1,745	$1,745	$1,745
				Aesthetic & Recreational			**$2,131**	**$1,745**
				Urban Green Space Total			**$2,473**	**$1,916**

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
MD	Multiattribute Decision Analysis
EA	Energy Analysis
MP	Marginal Product Estimation
CRS	Combined Revealed and Stated Preference
MA	Meta-anlaysis
VT	Value Transfer

Appendix D. Technical Appendix

Hedonic Model Specifications

Any regression equation is based on an assumed functional form, e.g., linear, quadratic, exponential, etc. Each of these functional forms implies a different type of relationship between independent question and home price. For instance, in regards to a variable for number of bedrooms: the sale price could increase in proportion to number of bedrooms, it could increase exponentially "without limit", it could increase but at a decreasing rate so that it "levels off" after five or six bedrooms, implying that the first few bedrooms are worth more on the margin, etc. Specifying the correct functional form not only can help avoid erroneously finding no relationship when one really exists or *vice versa*, but it can also help better characterize how a marginal change in an amenity affects price at various levels of that amenity.

The following are the functional forms used most commonly in hedonic analysis:

1. In a *linear* model, variables are expressed in terms of their absolute magnitudes, e.g., distance to park. In such an equation, a given coefficient can be thought of as the marginal change in price (measured in dollars) due to a one-unit change in that predictor variable, holding all else constant. (A similar interpretation holds for the less commonly used *quadratic* form where the variable might be, for example, the square of distance to park.)
2. In the commonly used *semi-log* model, the dependent variable is the logarithm of the sale price rather than the price itself. In this case, the coefficient of a predictor variable can be interpreted as the percentage change in the sale price due to a unit change in the predictor, e.g., the percentage by which the sale price changes as a result of being one unit distance closer to a beach or other environmental amenity. In this case the marginal implicit price of an amenity varies with the magnitude of sales price. For example, being 500 feet closer to a park may have a different percentage impact on the sale price of a million-dollar home than on the sale price of a $200,000 home. Likewise, having a fifth bathroom will add a different amount to two respectively priced
3. In a *log-log* or *trans-log* model, both the sale price and the predictor variables are expressed as logarithms ("logged"). Here the coefficient on a logged predictor variable can be interpreted as an "elasticity", that is, the percentage change in the sale price due to a one percent change in the predictor. In this case the marginal implicit price of an amenity varies with the magnitude of sales price and with the magnitude of the amenity. For example, a fifty foot change in a home's distance from a protected wetland may affect sale price differently if the change is from 50 to 150 fifty feet versus 500 to 550 feet or the addition of fifth bathroom may affect price differently than addition of a second bathroom; and in either of these cases, the percentage change in price may be different for more and less expensive homes.

The functional form and included variables of our hedonic equations were slightly different for each submarket. While each equation utilized the log-transformed dependent variable, the extent to which independent variables were transformed varied. The transformed dependent variable was chosen based on both the hedonic literature and on analyses of residual versus fit plots, which indicated nonlinearity in the relationships, and by the significant increase in R-squared due to transformation. Logging the dependent variable means that the coefficients of all linear independent variables can be interpreted as the percent change in the dependent variable due to a 1% increase in the independent variable. The coefficients of logged independent variables can be interpreted as elasticities.

The decision of which independent variables to transform—as well as which to include—was based on multi-model inferential statistical procedures {Burnham, 2002 #719}. This approach shows that minimization of the Akaike Information Criterion (AIC) can help select the "order" of likelihood of a set of nested or non-nested models. The commonly used measure of model fit, R-squared, is often not appropriate for comparison because it will always show the more complex model to be superior.

However, complexity comes at the tradeoff of parsimony, and therefore it is commonly accepted that a better model is one that increases fit relative to the number of parameters. AIC, on the other hand, penalizes models that are less parsimonious. By accounting for the tradeoff between model fit and complexity, it can show us which models best compromise between the two. The AIC is given by the equation is:

$$AIC = -2\log L(M) + 2k$$

Where:

k = the number of parameters plus one

log*L(M)* = the maximized log likelihood for the fitted mode

By comparing AIC scores of models including different independent variable combinations and transformations, we were able to derive a set of well fitting but parsimonious models. In general, many of the structural control variables, including lot area, living area, and improvement value were frequently log-transformed, while only a few of the distance control variables were.

Second-Stage Hedonic Analysis

The second stage seeks to estimate homeowners' demand curve for environmental amenities based on the hedonic price schedule derived from the regressions just mentioned. This function is based on consumer willingness to pay (WTP) for an attribute, which is not directly revealed. However, assuming individuals are price takers in equilibrium, a WTP function can be estimated from the marginal implicit attribute price (or "shadow price") derived in the first stage. Shadow prices are estimated by taking the partial derivative of price with respect to that amenity. The resulting equation describes how WTP for a marginal change in the amenity varies with quantity of or distance to the amenity. This is then solved for all distances or quantities observed. The resulting shadow price estimates are then regressed against distance/quantity.

Second stage hedonic analysis suffers from several major problems that are frequently cited in the literature. The most important is an econometric identification problem. This stems from the fact that the dependent variable in the second stage is not directly observed but is estimated from the hedonic price function, which is to say that both dependent and explanatory variables come from the same data source. This in fact can lead to getting the same parameters as in the first stage, in some cases (Mendelsohn, 1987). The identification problem also stems from the fact that price and quantity are chosen simultaneously by individuals.

One approach that has been used to deal with this problem is adding to the regression so-called "demand shifters," which are exogenous independent variables that at least partially correct for simultaneity (Mendelsohn, 1984). Frequently used demand shifters include socio-economic variables such as income and education levels. A more robust approach is the use of segmented housing markets to control for the identification problem (Freeman, 2003). This approach is superior because consumers in different markets with the same demand shifter characteristics (e.g. income) will face different marginal implicit attribute prices. Under this approach, separate hedonic equations are estimated for each market in the first stage (as we did for this study), yielding marginal implicit price estimates for each individual in each market, and in the second stage these values are regressed against the quantity of the attribute and some demand shifter.

Assuming a partial equilibrium analysis, where the magnitude of change in the resource quantity is small enough not to affect prices, and the time frame is relatively short run, the welfare value of a nonmarginal change in a resource can be determined for an individual by integrating under the WTP curve. In cases where the bounds of the non marginal quantity being assessed are large enough to affect price (e.g. the value of all wetlands in North America, where the two possible conditions are either all or none), a full equilibrium approach must be taken, which accounts for the endogeneity of price. Because potential non-marginal changes in the case of New Jersey are relatively small we assume that prices remain constant. The sum of the areas under the curve for all affected households then represent a lower bound of the welfare estimate (Bartik, 1988).

Landscape Modeling Framework

` The Landscape Modeling Framework (LMF) was designed to serve as a tool in integrated analysis of the interactions among physical and biological dynamics in a watershed, conditioned on socioeconomic behavior in the region. To account for ecological and economic processes in the same modeling framework we need to provide free exchange of information between the ecological and economic components. That immediately translates into the requirement that the scale and resolution of the spatial, temporal and structural interpretations are adequate to represent both of them. In particular, the spatial representation should be matched so that land use or land cover transformations in one component can be communicated to the other one. For such purposes it may be difficult to employ the more conventional approach based on spatial aggregation to larger units, called elementary landscapes, elementary watersheds, elementary areas of pollution or hillslopes (Beven and Kirby, 1979; Krysanova et al., 1989; Band et al., 1991; Sasowsky et al., 1991). These units are considered homogeneous and form the basis for the hydrologic flow network. The boundaries between spatial units are fixed and cannot be modified during the course of the simulation, which may be somewhat restrictive, if we are to consider scenarios of land use change, generated by the economic considerations, which were not envisioned in the design of the elementary spatial units.

A more mechanistic approach seems to be better suited to keep track of landuse changes and how they affect environmental conditions. We may present the landscape as a grid of relatively small homogeneous cells and run simulations for each cell with relatively simple rules for material fluxing among neighboring cells (Sklar et al., 1985; Burke et al., 1990; Costanza et al., 1990; Engel et al., 1993; Maxwell, 1995). This fairly straightforward approach requires extensive spatial data sets and high computational capabilities in terms of both data storage and calculation speed. However, it provides for quasi-continuous modifications of the landscape, where habitat boundaries may change in response to socioeconomic transformations. The LMF approach may be considered as an outgrowth of the approach first developed in the Coastal Ecosystem Landscape Spatial Simulation (CELSS) model (Sklar et al., 1985; Costanza et al., 1990), and later applied to a series of wetland areas, the Everglades clearly being the most sophisticated example (Fitz, in pressA; Fitz, in pressB).

The two main components of the LMF are the Spatial Modeling Environment (SME) and the Library of Hydro-Ecological Modules (LHEM). While SME is the computational engine that takes care of all input-output, data processing and number crunching, the LHEM provides the essential models and modules that actually describe the watershed and the ecological processes that occur there. The modular design of the LMF provides essential flexibility and transparency in model design and analysis.

We have used the LMF to construct the Hunting Creek Model (HCM) that we used in this study. The local dynamics in the HCM were similar to those developed in the Patuxent Landscape Model (Voinov et al., 1999), but the spatial implementation, defined by the Study Area, and the spatial resolution were different. By focusing on a relatively small watershed, we could make many more model runs, better calibrate the model, and refine our understanding of some of the crucial ecological processes and spatial flows in the ecosystem. The HCM became one of the most thoroughly calibrated and studied implementations under the LMF paradigm, and seems to be well-suited for the sensitivity experiments that we intend to undertake to understand how spatial allocation and processes in the watershed can influence ecosystem services and functions.

Model structure

The modeled landscape is partitioned into a spatial grid of unit cells. The model is hierarchical in structure, incorporating the ecosystem-level unit model for local, vertical dynamics that is replicated in each of the unit cells representing the landscape (Figure E1). With this approach, the model builds on the format of a raster-based geographic information system (GIS), which is used to store all the spatially referenced data included in the model. Thus, the model can be considered an extension of the analytical function of a GIS, adding dynamics and knowledge of ecological processes to the static snapshots stored in a GIS.

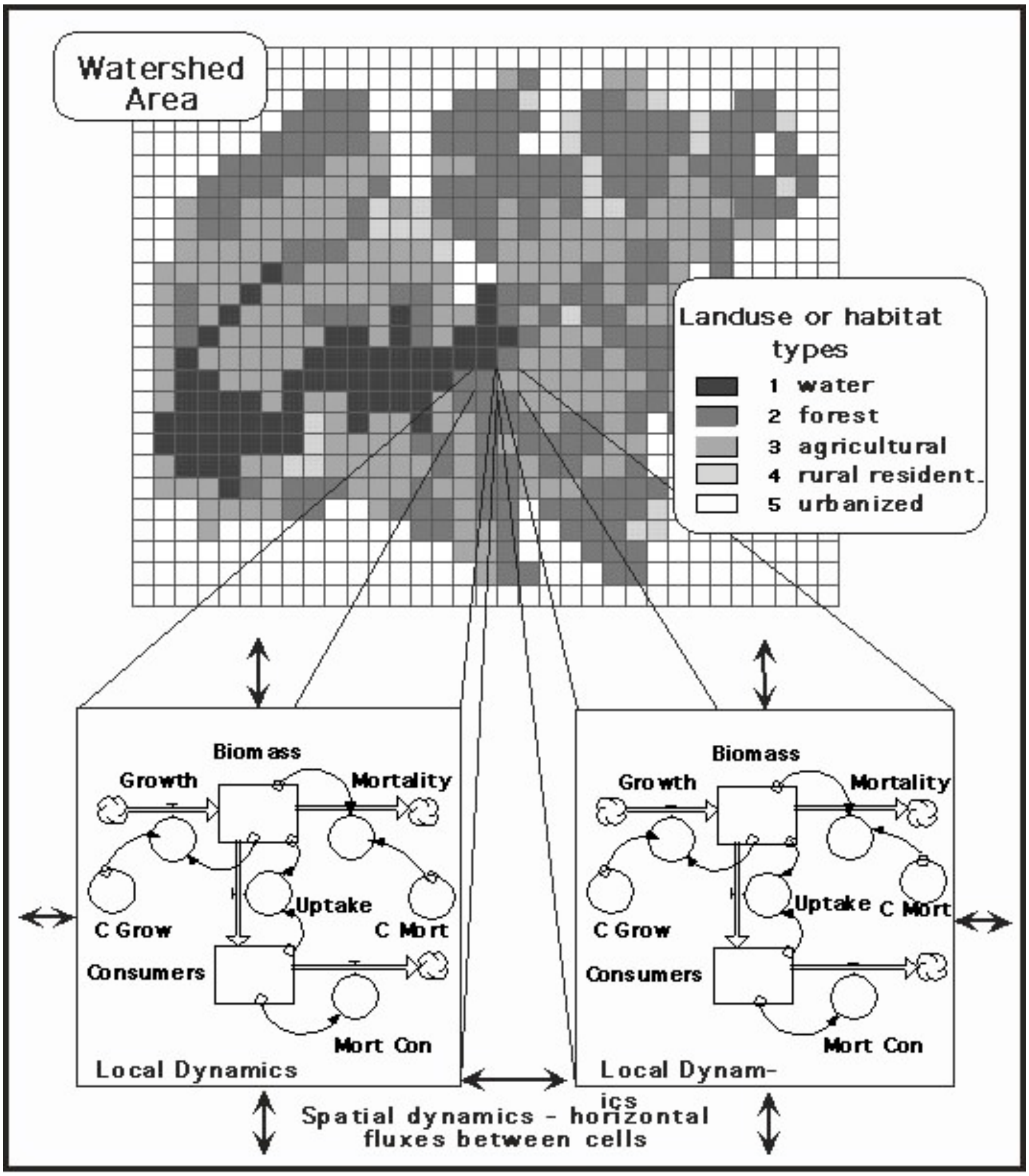


Figure E1: Spatial organization of the watershed model. Each cell is represented by a unit model.

Although the same local unit model runs in each cell, individual models are parameterized according to habitat type and geo-referenced information for a particular cell. The habitat dependent information is stored in a parameter database that includes initial conditions, rate parameters, stoichiometric ratios, etc. The habitat type and other location-dependent characteristics are referenced through links to GIS files. In this sense, the HCM is one of several site-specific ecological models that are process-based and are designed to apply to a range of habitats. The unit model in the HCM aims for an intermediate level of complexity so that it is flexible enough to be applied to a range of ecosystems but is not so cumbersome that it requires a supercomputer.

The unit models in each cell exchange matter and information across space. The horizontal fluxes that join the unit models together are defined by surface and subsurface hydrology. Alternative horizontal fluxes could be movement of air, animals, and energy such as fire and tidal waves although at this stage the HCM fluxes only water and entrained material. The spatial hydrology module calculates the amount of water fluxed over the surface and in the saturated sediment. The fluxes are driven by cell-to-cell head differences of surface water and saturated sediment water, respectively. Water fluxes between cells carry dissolved and suspended material. At each time step, first the unit model updates the stocks within each cell due to vertical fluxing and then cells communicate to flux matter horizontally, simulating flows and determining ecological condition across the landscape.

Figure E2 shows how the various modeled events are distributed in time when simulated in the HCM. The model employs a time-step of 1 day, so most of the ecological variables are updated daily. However, certain processes can be run at longer or shorter time intervals. For example some spatial hydrologic functions may need an hourly time step, whereas certain external forcing functions are updated on a monthly or yearly basis.

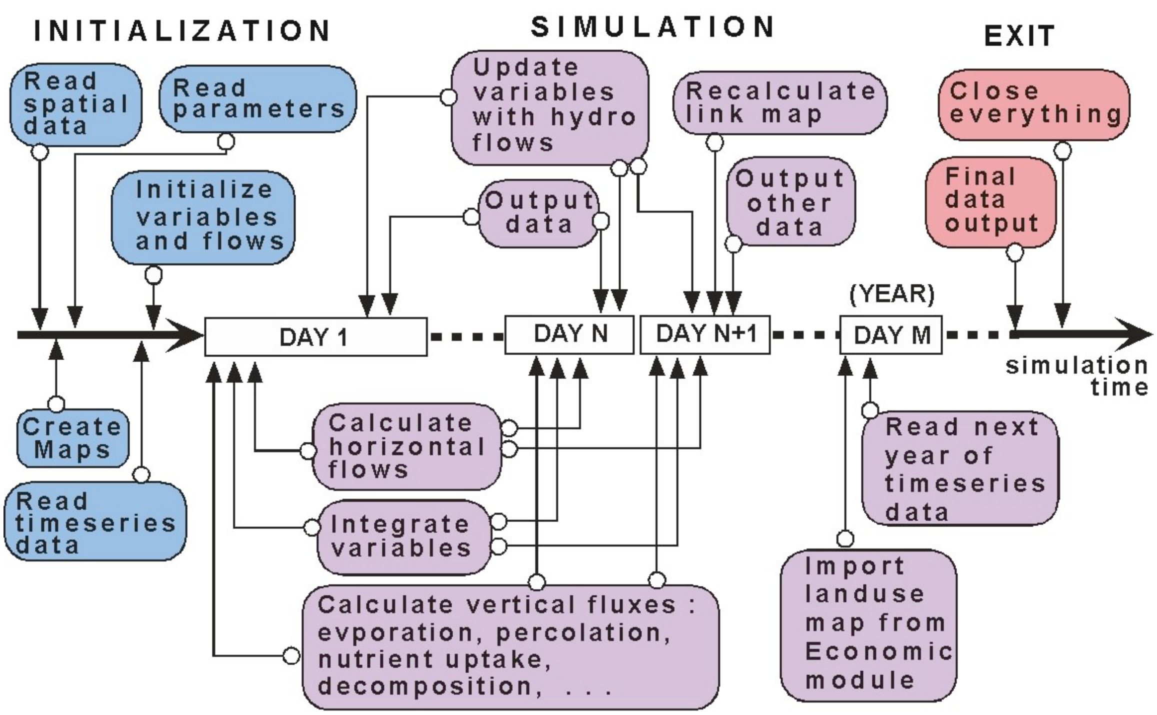


Figure E2: During simulation in HCM, model events are distributed in time.

The LHEM was used as a source of local (run in each cell) and spatial modules (run over many cells to present horizontal movement of material). Modules were picked from the Library to represent hydrology, nutrient movement and cycling, terrestrial primary productivity, and dynamics of organic decomposition (Figure E3). The hydrology module of the unit model is fundamental to modeled processes since it links the climatic forcing functions to chemical and biotic processes, and allows feedbacks between sectors. Phosphorus and nitrogen are cycled through plant uptake and organic matter decomposition, with the latter simulated in the sector that describes the sediment/soil dynamics. The module for plants includes growth response to various environmental constraints (including water and nutrient availability), changes in leaf canopy structure (influencing water transpiration), mortality, and other basic plant dynamics. Feedbacks among the biological, chemical and physical model components, influence ecosystem response to changing conditions. For further details on LMF go to the web site at http://giee.uvm.edu/IDEAS.

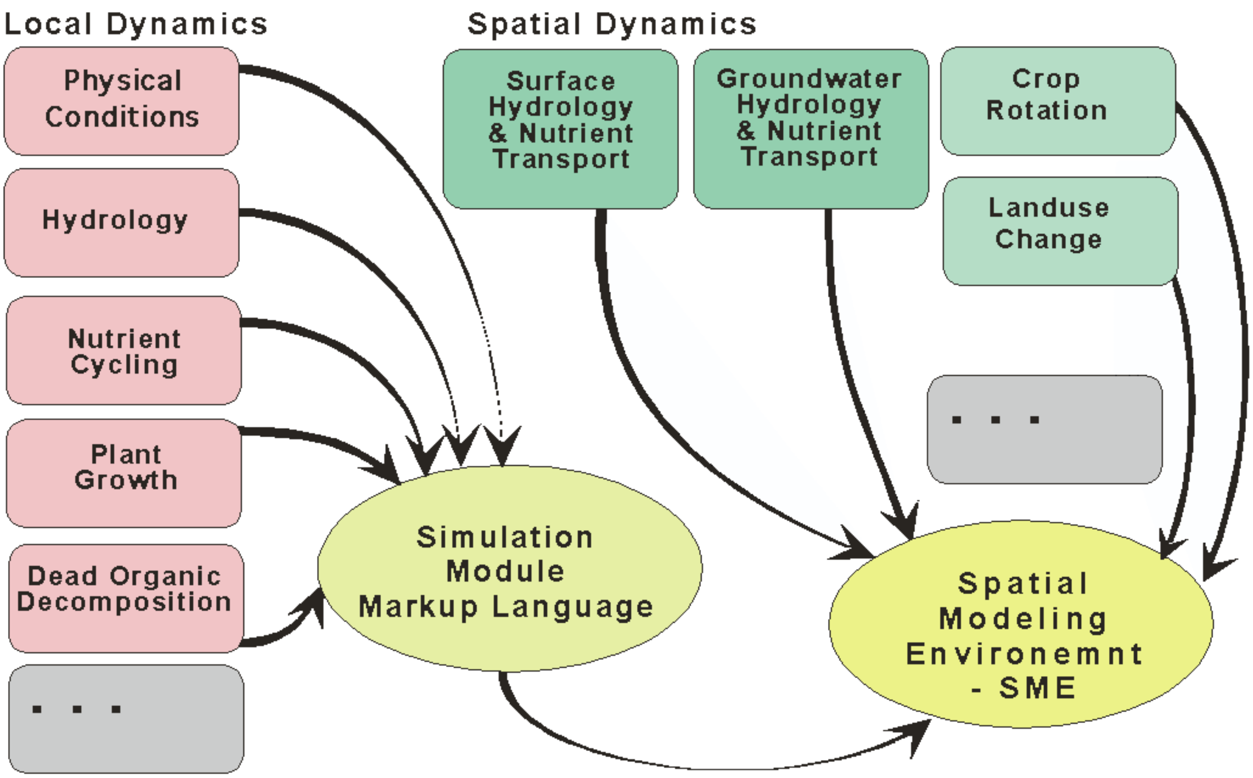


Figure E3: Flow diagram of model.

Hydrology

The traditional scheme of vertical water movement (Novotny and Olem, 1994), assumes that water is fluxed along the following 4-phase pathway: rainfall -> surface water -> water in the unsaturated layer -> water in the saturated zone. Snow is another storage that is important to mimic the delayed response caused by certain climatic conditions. In each of the stages, some portions of water are diverted due to physical (evaporation, runoff) and biological (transpiration) processes, but in the vertical dimension the flow is controlled by the exchange between these 4 major phases. Taking into account the temporal (1 day) and spatial (200 m) resolution of the HCM formalization and of the available input data, we can simplify this model as follows.

At a daily time step, the model cannot attempt to mimic the behavior of shorter-term events such as the fast dynamics of a wetting front, when rainwater infiltrates into soil and then travels through the unsaturated zone towards the saturated groundwater. During a rapid rainfall event, surface water may accumulate in pools and litter-fall but in a catchment such as the Hunting Creek watershed, over the period of a day, most of this water will either infiltrate, evaporate, or be removed by horizontal runoff. Infiltration rates based on soil type within the Patuxent and Hunting Creek watersheds range from 0.15 to

6.2 m/day (Maryland Department of State Planning, 1973), potentially accommodating all but the most intense rainfall events in vegetated areas. The intensity of rainfall events can strongly influence runoff generation, but climatic data are rarely available for shorter than daily time steps. Also, if the model is intended to run over large areas for many years, the diel rainfall data become inappropriate and difficult to project for scenario runs. Therefore, a certain amount of detail must be forfeited to facilitate regional model implementation.

With these limitations in mind, we assume that rainfall infiltrates immediately to the unsaturated layer and only accumulates as surface water if the unsaturated layer becomes saturated or if the daily infiltration rate is exceeded. Ice and snow may still accumulate. Surface water may be present in cells only if it is in rivers, creeks, lakes, and ponds. Sheet surface water is removed by horizontal runoff or evaporation. Within the daily time step, surface water flux will also account for the shallow subsurface fluxes that bring the water distributed over the landscape into the micro channels and eventually to the river. Thus, the surface water transport takes into account the shallow subsurface flow that may occur during rainfall, allowing the model to account for the significantly different nutrient transport capabilities between shallow and deep subsurface flow.

Conceptually this is close to the slow and quick flow separation (Jakeman et al., 1993; Post and Jakeman, 1996) assumed in empirical models of runoff. In our case the surface water variable accounts for the quick runoff, while the saturated storage performs as the slow runoff, defining the base-flow rate between rainfall events.

Nutrients

In LHEM, the nutrients considered are nitrogen and phosphorus. Various nitrogen forms, N02-, N03- and NH4+ are aggregated into one variable representing all forms of nitrogen that are directly available for plant uptake. Available inorganic phosphorus is simulated as orthophosphate. The distinction between N and P cycles appears in conceptualizing nutrients on the surface, since in the model they are no longer associated with surface water and therefore need not be in the dissolved form. On the contrary, since most of the time most of the cells have no surface water, N_S (P_S) represents the dry deposition of nitrogen (phosphorus) on the surface. Over dry periods N_S (P_S) continues to accumulate with incoming fluxes from air deposition or mineralization of organic material. When rainfall occurs, a certain proportion of the accumulated N_S (P_S) becomes dissolved and therefore is made available for horizontal fluxing and infiltration.

Further modification of the nutrient dynamics was required to accommodate the aggregation of surface and shallow subsurface flows in the hydrologic sector. In the PLM a proportion of nitrogen and phosphorus stored in the upper soil layer is made available for fast horizontal fluxing along with nutrients on the land surface. We have assumed this layer to be 10 cm thick, following a similar formalization in the CNS model (Haith et al., 1984), where this upper soil layer was also assumed to be exposed to direct surface runoff.

In addition to N_S (P_S), mineral N (or P) on the surface, and N_{SD} (P_{SD}), mineral N (or P) in the sediment, the phosphorus cycle features another variable P_{SS}, which is the phosphorus deposited in the sediment in particulate form, no longer available for plants uptake, and effectively removed from the phosphorus cycle. At higher concentrations the dissolved PO, becomes absorbed by the organic material and metal ions in the soil. Therefore the rate of sorption is also controlled by the amount of organic material in the soil, which in this case mostly consists of soil microorganisms (microbes). At lower concentrations of soluble PO_4 in the sediment, P_{SS} becomes available again and returns to the cycle.

Plants

The LHEM plants module includes dynamics in carbon-to-nutrient ratios that are important to woody and perennial plant communities (Vitousek et al., 1988) and introduces important differences between evergreen and deciduous plant communities. Additional fluxes were added to allow for human intervention through fertilizing, planting and harvesting of crops and trees. The newly revised macrophyte sector can now simulate the nutrient storage of a forest ecosystem in multiple year simulations and allow scenarios for Best Management Practices (BMP's) in agriculture and urban lawns.

Plants are represented by two state variables for photosynthetic and non-photosynthetic plant matter. The carbon to nutrient ratios (C:N:P ratios) for both state variables link to different steps in the N and P nutrient cycles. The C:N:P ratio in the photosynthetic part of the nutrient cycles is instrumental in controlling uptake and the resulting accumulation of organic nitrogen and phosphorous. The C:N:P ratio in the non-photosynthetic biomass is used to estimate the rates of decomposition and the extent of nutrient mineralization. The C:N:P ratios tend to increase as woody biomass low in nutrient content accumulates in aging forests. Our strategy still assigns fixed C:N:P ratios to the photosynthetic biomass, but relates changes in the non-photosynthetic biomass C:N:P ratios to changes in woody biomass, bringing estimated nutrient storages closer to measured values.

Some concepts were redefined in the new model to represent a greater variety of habitats. The terms evergreen and deciduous are broadly interpreted to encompass not only trees but other plant communities as well. Most of the agricultural crops and annual herbs are considered deciduous, while wetlands, grasslands and lawns are considered evergreen. The main difference between the deciduous and non-deciduous plant communities is that a fall hormonal trigger mechanism causes the deciduous plants to shed the photosynthetic part of the plant, while recovering some of the biomass for the non-photosynthetic tissues. No recovery of biomass occurs from leaf mortality. It is during this fall period when seeds and tubers are formed and photosynthetic products are stored in tree root systems. In the spring deciduous plants experience accelerated growth in addition to a seasonal growth also experienced in the evergreen community.

Allocation of photosynthetic products to leafy or woody tissues is controlled by the maximum in the ratio of photosynthetic to non-photosynthetic materials (*Max-ph:nph*). An accelerated spring growth, simulating sap flow in trees and seed germination, was introduced for the deciduous portion of the plant community. Labile carbon stored in non-photosynthetic tissues (roots, stems and branches) is translocated to produce photosynthetic tissue (leaves) in an attempt to reach a community-specific *Max-ph:nph*. Translocation from the non-photosynthetic tissue to the photosynthetic tissue comes to a halt when all labile carbon is used from storage, or the *Max-ph:nph* ratio is reached, or hormonal activity ceases. New photosynthetic products are created in the leaves, under the various environmental restrictions. These newly available products can be allocated to additional leaf growth if *Max-ph:nph* is not yet reached, or can be translocated back to the non-photosynthetic parts for growth of woody matter or storage. Growth in woody matter offsets the photosynthetic to non-photosynthetic ratio from *Max-ph:nph* and allows for additional growth in leafy material.

Detritus

At present this module serves predominantly to close the nutrient and material cycles in the system, it does not go into all the details of the multi-scale and complex processes of leaching, bacterial decomposition, etc. As biomass dies off, a part of it turns into Stable Detritus, D_S, whereas the rest becomes Labile Detritus, D_L. The proportions between the two are driven by the lignin content, which is relatively low for the PH biomass and is quite high for NPH biomass. Labile detritus is decomposed directly, and stable detritus is decomposed either to labile detritus, or becomes Deposited Organic Material (DOM), D_{DOM}.

Avoiding many of the complexities, we assume that the decomposition process is linear for the decay of Stable Detritus as follows:

$$F_{DS} = d_0 \cdot D_S + d_1 \cdot L_{DT} \cdot D_S,$$

where d_0 is the flow rate of stable detritus transformation into D_{DOM} and d_1 is the flow rate between stable and labile detritus. The latter flow is modified by the Vant-Hoff temperature limitation function $L_{DT} = 2^{(T-20)/10}$, where T is the ambient air temperature (^oC). The decomposition of Labile Detritus and DOM is described similarly as linear functions modified by the Vant-Hoff temperature function.

Spatial implementation

Once the local ecological processes were described, we needed to decide on the algorithms that put the local dynamics within a spatial context. For watersheds in general and for Hunting Creek in particular, hydrologic fluxes seem to be the most important mechanism linking the cells together and delivering the suspended and dissolved matter across the landscape.

The importance of hydrologic transport has been long recognized and considerable effort has been put into creating adequate models for various landscapes (Beven and Kirby, 1979; Beasley and Huggins, 1980; Grayson et al., 1992). Nevertheless there are no off-the-shelf universal models that can be easily adapted for a wide range of applications. As part of a more complicated modeling structure, the hydrologic module is required to be simple enough to run within the framework of the integrated physical-ecological model yet sufficiently detailed to incorporate locally important processes. As a result, some hydrologic details need to be sacrificed to make the whole task more feasible, and these details may differ from one application to another, depending upon the size of the study area, the physical characteristics of the slope and surface, and the goals and priorities of the modeling effort.

To simplify hydrologic calculations, we merge process-based and quasi-empirical algorithms (Voinov et al., 1998). First, given the cell size within the model (200 m), every cell is assumed to have a stream or depression where surface water can accumulate. Therefore the whole area becomes a linked network of channels, where each cell contains a channel reach which discharges into a single adjacent channel reach along the elevation gradient. An algorithm generates the channel network from a link map, which connects each cell with its one downstream neighbor chosen from the eight possible nearest neighbors.

Second, since most of the landscape is characterized by an elevation gradient, the flow is assumed to be unidirectional, fluxing water down the gradient. In the simplified algorithm, a portion of water is taken out of a cell and added to the next one linked to it downstream. To comply with the Courant condition (Chow et al., 1988), this operation is reiterated many (10-20) times a day, effectively generating a smaller time step to allow faster runoff. The number of these iterations was calibrated so that the water flow rates match gage data.

This procedure was further simplified by allowing the water to flow through more than one cell over one iteration and then generalized by assuming a variable number cells in the downstream link, as a function of the amount of water in the donor cell. This was adopted to allow for a faster flow when more water is available on the surface (Voinov et al., 1999). It increased flexibility in describing individual hydrographs and in generalizing them over longer time periods and over larger watershed areas.

For groundwater movement we used a linear Darcy approximation that moves water among adjacent cells in proportion to a conductivity coefficient and the head difference. The groundwater movement provides the slow water flow that generates the river base-flow. Surface water runoff is the major determinant of the peak flow observed.

Hunting Creek data

Spatial hydrologic modeling requires extensive data sets. Most of the spatial coverages for the HCM were derived from the data sets previously assembled for the whole Patuxent watershed. In Fig. 6 we present the basic spatial coverages that have been employed in our modeling effort and some of the derived layers that were also essential for the hydrologic module. Spatial fluxes of surface water in watershed models are predominantly driven by the elevation gradient. In this study we used the United States Geological Survey's (USGS) Digital Elevation Model (DEM) data that are available for downloading from the Internet (USGS, 1995).

USGS offers elevation data in 1 degree grid coverages for the 4 map quadrangles covering the Patuxent watershed. DEM grids are based on 1:250,000 USGS maps with 3-arc second grid spacing. Grids constructed from USGS I-degree DEMs are not immediately suitable for the analysis of such topographic features as volume, slope, or accurate visibility, because they measure the x, y (planar) locations as latitude and longitude, while the z value (elevation height) is measured in meters. Consequently, the actual distance on the ground represented by one ground unit is not constant, and the ground distance units and the surface elevation units are not the same. To make this surface model compatible with other layers of information and suitable for analysis, the ground units in the I-degree USGS DEM have been projected into non-angular units of measure such as the LTTM coordinates. After reprojection, the grid was rescaled to the 200 m resolution, which is the highest resolution currently used in the PLM. The vertical resolution of the DEM maps is 1 m.

Using a GIS the DEM data have been preprocessed to create several other raster maps needed for the hydrologic model. Watershed Boundary (Study Area map), Slopes (Fig.7) and Aspects layers have been calculated by the Watershed Basin Analysis Program in GRASS - Geographic Resources Analysis Support System (USACERL, 1993).

The River Network coverage (Figure E4) has been acquired from the TIGER/LINE database (USCB, 1996) in a vector format. The database contained numerous errors: streams that were not continuous, missing channels (improperly digitized or missing on the original maps or photos because they may have been dry at the time the photos/maps were interpreted). The hydrologic analysis tools in the ARC/INFO GRID module (ESRI, 1994) were applied to correct the digitized stream network. Using the digital elevation model as an input we delineated the drainage system and then quantified its characteristics. For any location in the grid, those tools also gave us the upslope area contributing to that point and the downslope path water would follow. A "hydrologically proper" surface, without any artificial pits or hills, was produced and flow directions and flow accumulations were determined. Water channels were identified for different threshold amounts of water accumulation (product of the number of cells draining into a target cell and the size of the precipitation event). These water channels were used as a background coverage to manually correct stream discontinuities for the digitized River Network. The corrected River Network was converted into a raster (cell -based) format in order to comply with other data layers. This River Network map produced from the elevation data turned out to be more consistent, than the original vector map.

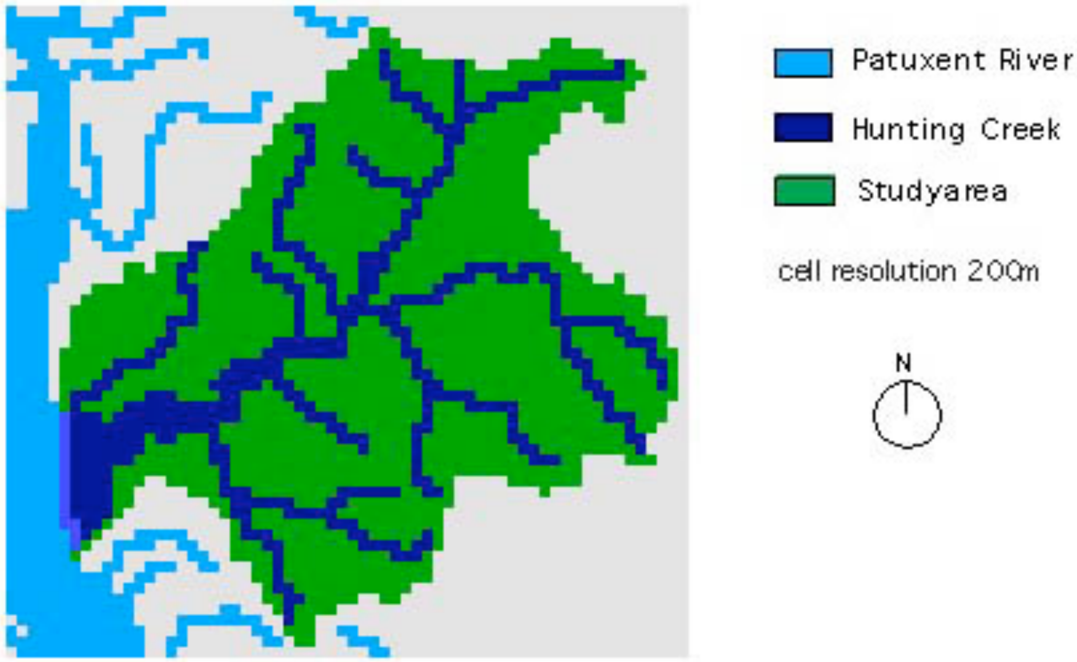


Figure E4: River network for Hunting Creek watershed based on TIGER database. data.

The Soils layer was originally imported from the State Soil Geographic (STATSGO) data base (NRCS USDA, 1997) which has been compiled using a USGS 1:250,000 scale, 1 by 2 degree quadrangle series as a map base. The STATSGO Data Base was downloaded in GRASS format and reprojected from the Albers Equal Area Projection to the needed LTTM projection. Every map unit on a STATSGO coverage contains up to 21 components (segments) for which there are attribute data. One of the disadvantages of this data set is that these components cannot be spatially identified, which reduces the STATSGO application to the coarse regional scale.

After we analyzed the tabular information it was clear that aggregation criteria did not include hydrological properties, because one map unit could contain soils from very different hydrological groups. Therefore we could use only some general hydrological parameters from STATSGO, but most of the spatially explicit soil data was taken from the Patuxent Watershed Counties Soils map (Figure E5) available from the Maryland Office of Planning (MOP) (Maryland Department of State Planning, 1973). The Groundwater Table Map, required as an initial condition for the model, was approximated from a series of spatial and point data sets using the GRASS overlay and interpolation techniques. The reference points were taken from:

- MOP Soils map and the unsaturated depth data that was provided by the Maryland Department of State Planning;
- the elevation and river network coverages, along which the groundwater table was assumed to reach the surface;
- 15 well measurements of the groundwater level over the watershed area (James, et al., 1990).

The groundwater depth data were interpolated over the whole watershed with these data sets as reference points. After that the model was run for 100 days, the Groundwater Table Map was regenerated, saved and then fed back into the model for subsequent runs as the initial condition for the depth of the water table. This improved the performance of the Hydrological Module by significantly decreasing the initial adjustment period in the model runs.

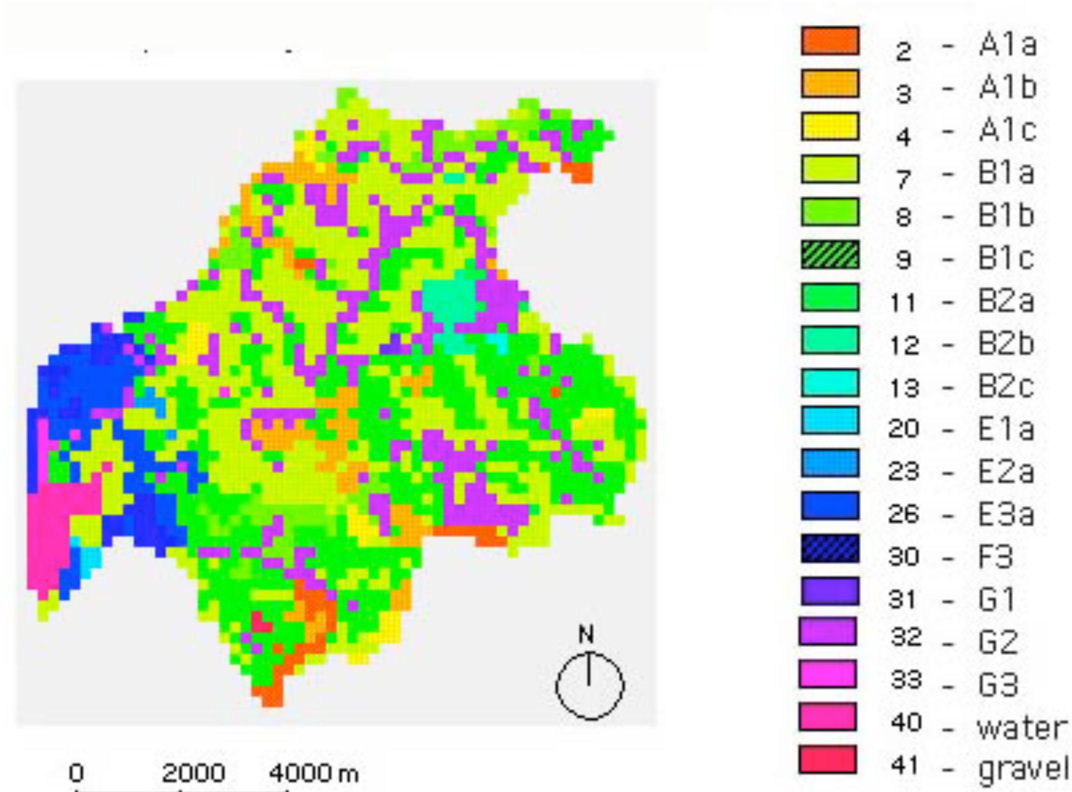


Figure E5: Soils for the Hunting Creek watershed based on Natural Soil Groups of Maryland (Table D1).

Table 1. Estimated physical and chemical properties of Natural Soils Grounp of Maryland (Maryland DePartment of state Planning, 1973)

GROUP	DEPSED	DEPWAT	DEPSOL	EROK	HYDGRP	IRRMAX	PERMAX	PERC	AWC	PH	TEXTUR
110 - 113	72+	4+	0 - 60	0.17	1	1	>6.0	<45	.02-.06	4.0 - 5.0	Lmy sd, sd sdy Lm
120	72+	1 - 10	0 - 60	0.17	1	N/A	>6.0	<45	<.06	3.0 - 8.0	Sand
210 - 213	72+	3+	0 - 60	0.32	2	0.4 - 0.6	0.60 - 2.0	45 - 60	.12 - .24	4.5 - 6.5	St lm, Lm, fine sdy Lm, sdy Lm, sty cy Lm, cy Lm, sty cy, cy
220 - 223	72+	4+	0 - 60	0.43	3	0.3 - 0.4	0.20 - 0.60	>60	.12 - .24	4.5 - 7.3	Silt loam, Loam, gravelty Loam, clay Loam, silty clay Loam
230	72+	5+	0 - 60	0.37	3	0.3	<.60	>60	.06 - .24	4.0 - 5.0	Clay, silty clay, silt Loam, Loam, Loamy sand
310 - 313	20 - 40	In Bedrock	0 - 40	0.22	3	0.3	0.60 - 6.0	<60	.12 - .24	4.0 - 7.3	St Lm, Lm, sly sty lm, sly Lm, chy Loam, chy st Lm, sdy Loam
320	20 - 40	3+	0 - 40	0.37	3	0.3	<0.60	>60	.12 - .24	5.0 - 7.5	Silty clam Loam, silty clay, clay
410 - 413	<20	In Bedrock	0 - 20	0.28	3 - 4	0.3	0.60 - 6.0	<45 - >60	.18 - .24	4.0 - 7.3	Shaly silt Loam, shaly Loam, silty clay Loam, silty clay
510 - 512	72+	1.5 - 2.5	0 - 60	0.28	3	0.4 - 0.6	0.60 - 6.0	<60	.12 - .24	4.0 - 5.0	Sandy Loam, sandy clay, Loam, Loamy sand, sand
520 - 522	72+	1 - 3	0 - 60	0.43	3	0.3 - 0.4	<.60	>60	.12 - .24	4.0 - 6.5	Silt Loam, Loam, silty clay Lm, fine sandy Loam, sdy clay Loam
530 - 532	72+	1.5 - 2.5	0 - 60	0.37	3	0.4	0.20 - 0.60	>60	.18 - .24	4.5 - 5.5	Silt Loam, Loam, silty clay Loam
610	72+	0 - 1	0 - 60	N/A	4	1	>6.0	<45	<0.06	3.5 - 5.0	Loamy sand, sand
620	72+	0 - 1	0 - 60	0.28	4	0.4 - 0.6	0.60 - 2.0	<60	.12 - .24	4.0 - 5.0	Sdy Loam, fine sdy Loam, sandy clay Loam, Loam, Loamy sand
630	72+	0 - 1	0 - 60	0.43	4	0.3	<0.60	>60	.18 - .24	4.0 - 7.8	Silty clay Loam, silty clay, clay, Loam, silt Loam
710 - 711	72+	3+	0 - 60	N/A	2 - 3	0.5 - 0.7	0.20 - 2.0	<45 - >60	.12 - .24	4.0 - 7.3	Silt Lm, Lm, fine sdy Lm, sdy Lm, sandy Loam, silty clay Loam
720	72+	0 - 1	0 - 60	N/A	4	0.5	0.60 - 6.0	<45 - >60	.18 - .24	4.0 - 7.3	St Lm, sty cy Lm, sty cy, fine sdy Lm, sandy Loam, Loam, muck
730	72+	0	0 - 60	N/A	N/A	N/A	Var	Var	Var	3.5 - 9.0	variable
810 - 813	Too variable to rate. Determine the specific soil series name from the detailed soil map and use the information for the group that the series is in.										
820 - 823	Too variable to rate. Determine the specific soil series name from the detailed soil map and use the information for the group that the series is in.										

Explanation

GROUP Natural Soil Group Code
DEPBED = Depth to bedrock (in.) -- distance from the surface of the soil downward to the surface of the rock layers. Soils were observed only to a depth of 6 feet; greater depths are specified as 72+ in.
DEPWAT = Depth of water table (ft.) -- distance from the surface of the soil downward to the highest level reached in most years by ground water.
DEPSOL = Soil depth (in.) -- this does not imply that the soils are only 60 in. deep, but rather that the estimates in the table are for th 0 - 60 in. depth and not below.
EROK = Erodibility (K factor) -- a measure of the susceptibility of the bare soil to erosion and the same K factor as that used in the Universal Soil Loss Equation (Wischmeier and Smith, 1965).
HYDGRP = Hydrologic Soil Group -- a measure of the runoff potential of soils when fully saturated. Group "A" soils have the lowest potential and "D" soils the highest.
IRRMAX = Maximum irrigation rate (in/hr) maximum rate of irrigation water applied by sprinklers.
PERMAX = Permeability (in/hr) -- rate at which soil transmits water while saturated. Permeability rates shown are based on the least permeable section of soil.
PERC = Percolation (min/hr) -- rate at which water can move through a soil with moisture at field capacity.
AWC = Available water Capacity (in/in) -- the difference between the amount of water in the soil at field capacity and the amount in the soil at the wilting point of most crops
PH = Reaction (pH) -- the degree of acidity or alkalinity of a soil group, expressed in pH units.
TEXTUR = Dominant texture -- relative percents of sand, silt, and clay in a soil sample. If the soil contains gravel or other particles coarser than sand, then a modifier is added.
Abbreviations from TEXTUR column: sd = sand, sdy = sandy, st = silt, sty = silty, cy = clay, lm = loam, chy = channery, shy = shaly,

Land Use 1990 Anderson II classification coverages (Figure E6) have been acquired from the Maryland Office of Planning in a vector format and then rasterized for the required cell resolution. In order to simplify the model and match the available sets of ecological parameters, the landuse was aggregated to 5 types. The aggregated version of the land use data (Figure E7) was developed using the algorithm described in Figure E8.

The climatic data series were taken from the EarthInfo Inc. NCDC Summary of the Day database (EARTHINFO, 1993). The point time series for Precipitation, Temperature, Humidity and Wind were then interpolated across the study area to create spatial climatic coverages. The calibration procedures were mostly based on USGS gaging data also available for downloading from the Web (USGS, 1995a). Most of the calibration runs were based on the gaging station located on Hunting Creek under the bridge on MD Route 263 approximately 2.4 miles South of Huntingtown. For this station we have data for the time period that matches the one defined by the climatic data series, that is 1990-1996.

The Calvert County Department of Planning and Zoning has provided the necessary zoning maps (Figure E9) that were important for generating the scenarios of land use change in the area. In addition the sewer planning maps (Figure E10) and maps of dwelling units densities (Figures D11 and D12) were provided by the same source.

Nitrogen fertilizer application for the farmlands of Calvert County (Table D2) has been calculated based on:

A. Natural Soils Group information (MOP),

B. Soil surveys for Calvert County;

C. MD's agronomical soil capability assessment program (defining yield expectations)

D. Plant nutrient recommendations based on soil tests and yield goals.

A nitrogen fertilizer application map has been developed using GIS techniques on the basis of soils and 1990 land use coverages (Figure E13) or on the basis of soils and projected land use scenarios (Figure E14).

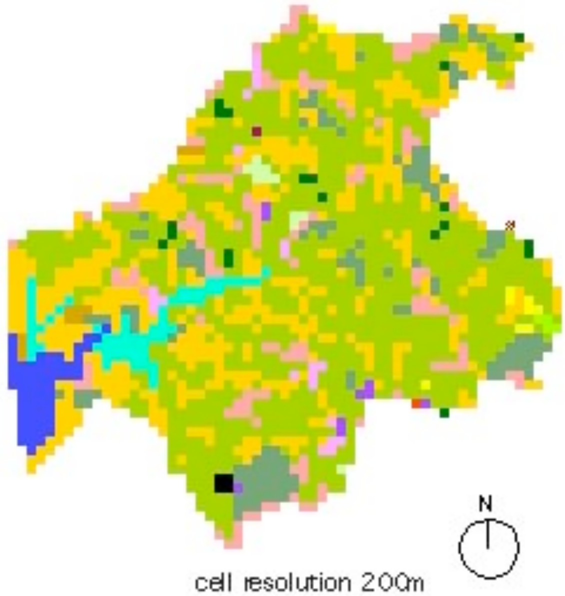


Figure E6: Landuse 1990 based on Maryland Office of planning data

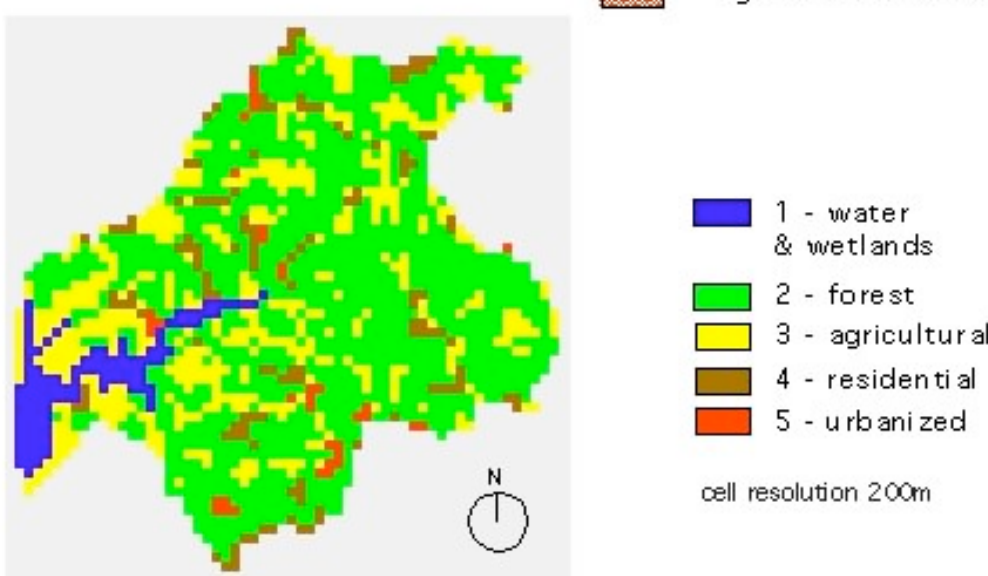


Figure E7: Aggregation level II of landuse 1990

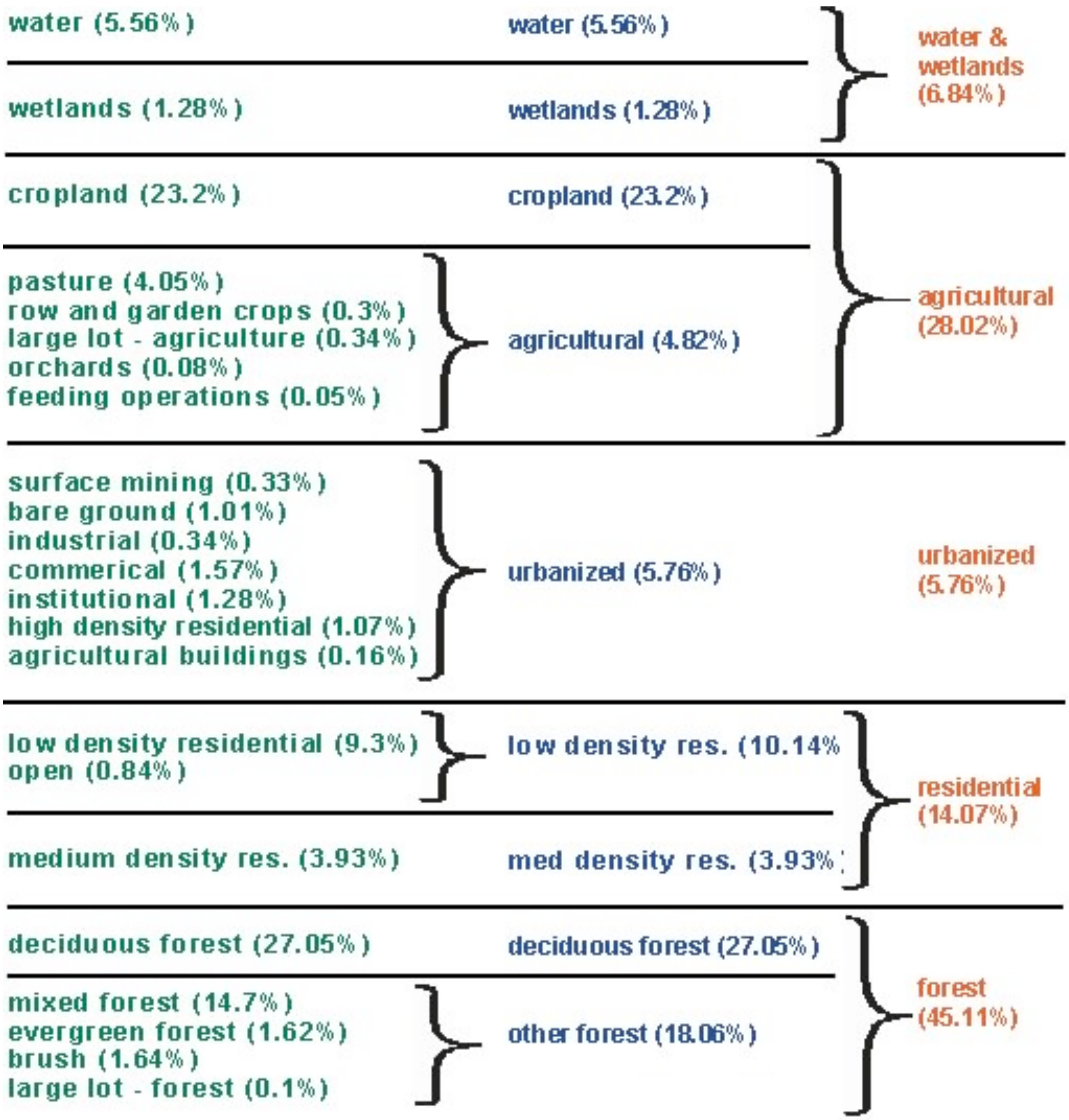


Figure E8: Aggregation of landuses assumed in the model.

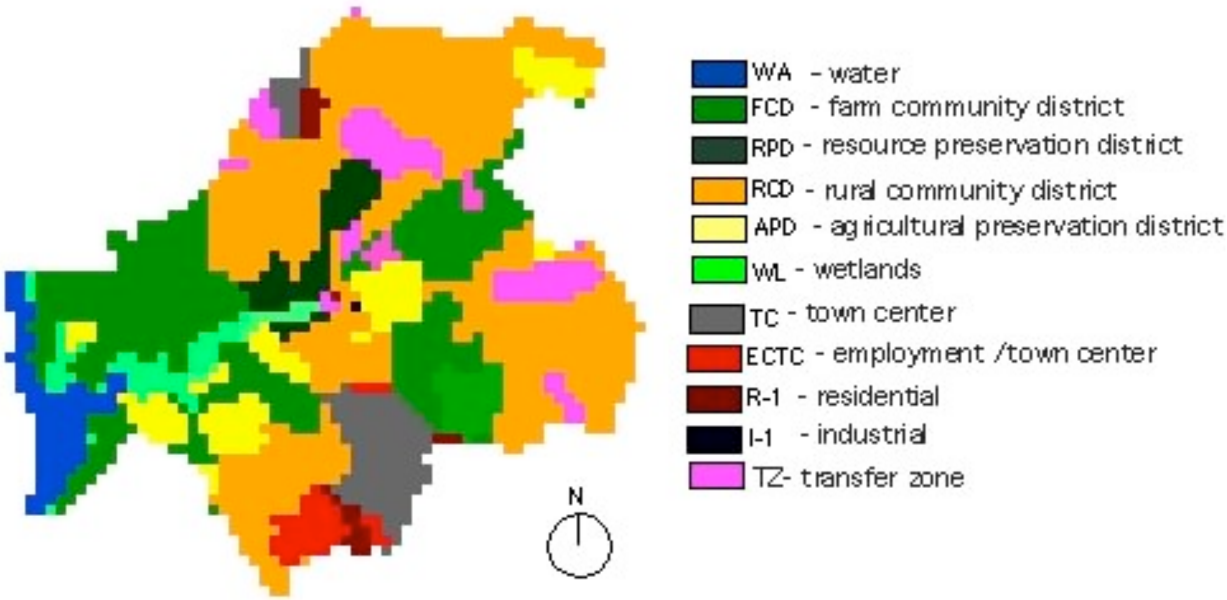


Figure E9. Zoning for Hunting Creek watershed based on data from Calvert County Department of Planning and Zoning.

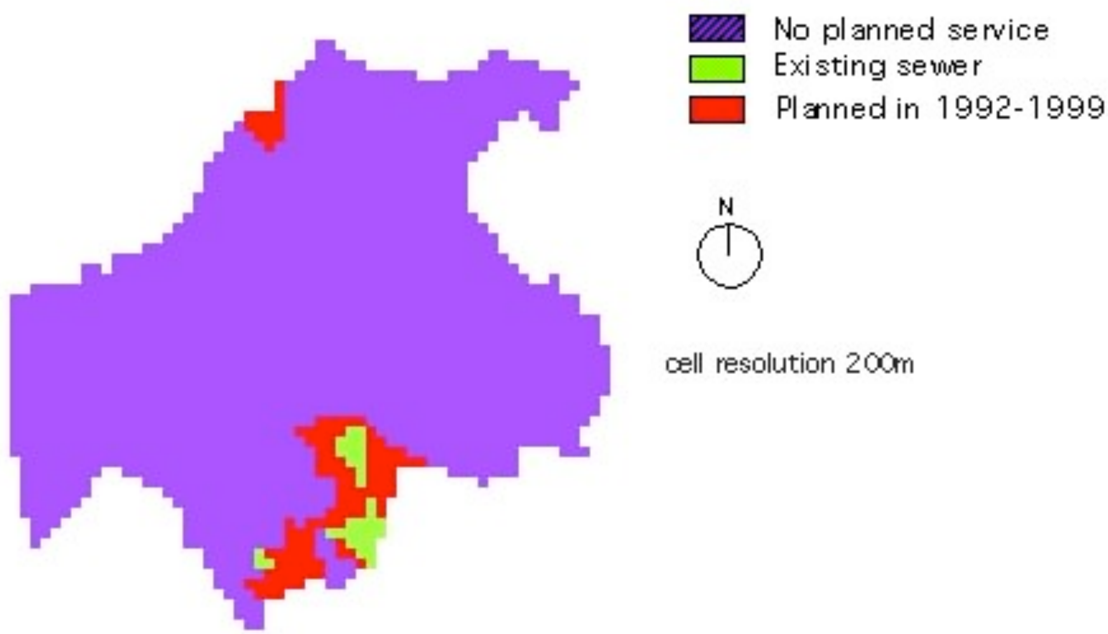


Figure E10. Zoning for Hunting Creek watershed based on data from Calvert County Department of Planning and Zoning.

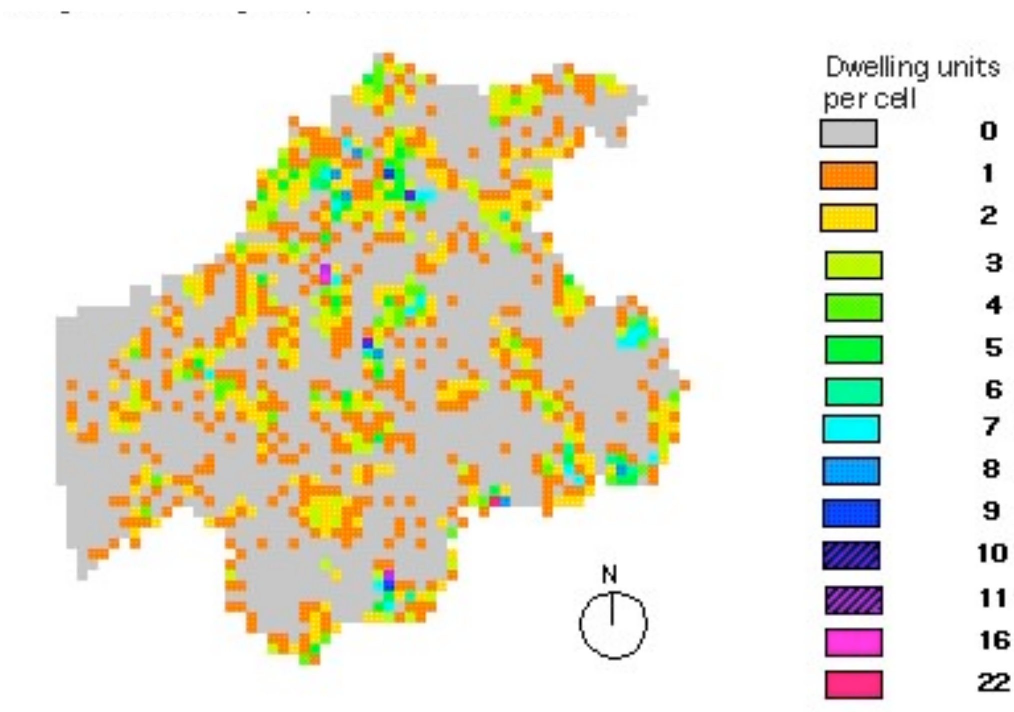


Figure E11: Density per cell (10 acres) of existing dwelling units for Hunting Creek watershed based on Calvery County Planning and Zoning Department Information.

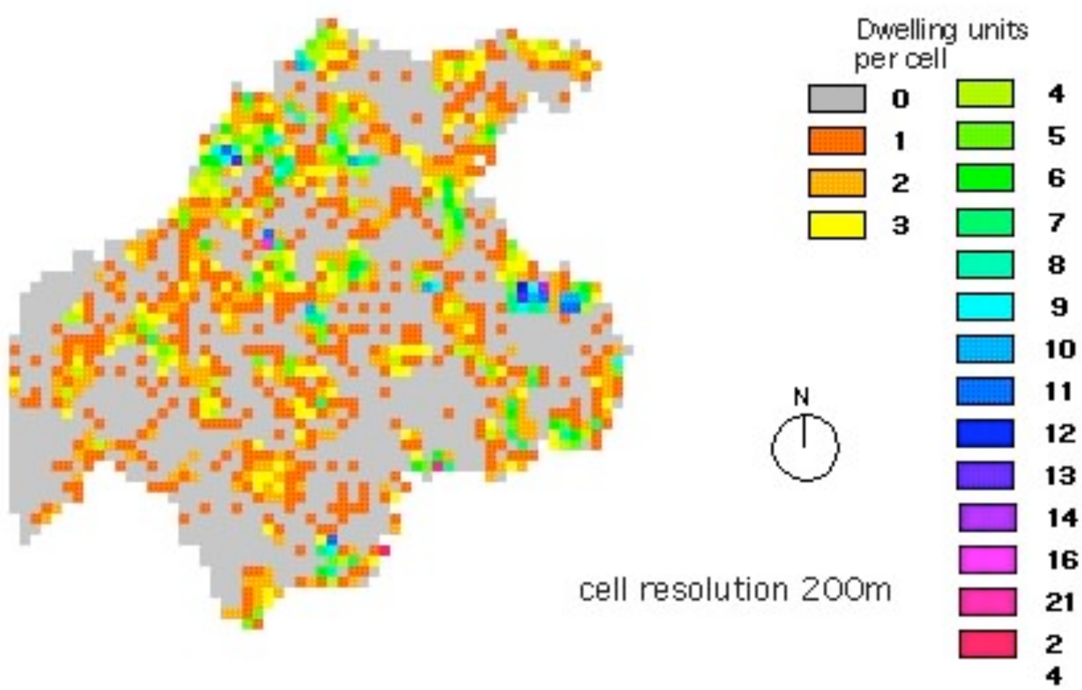


Figure E12: Density per cell (10 acres) of improved and unimproved dwelling lots for Hunting Creek watershed based on Calvert County Planning and Zoning Department Information.

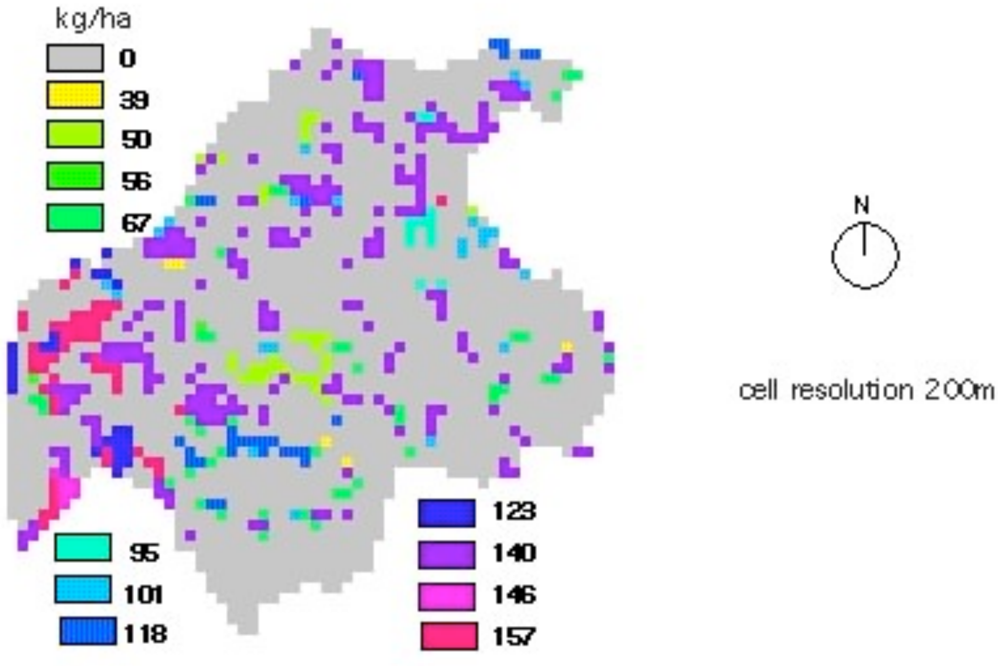


Figure E13: Estimated Nitrogen fertilizers applications (Table D2) based on Landuse 90 (Figure E7), Natural Soil Groups classification, corn yield expectation and plant nutrient recommendation.

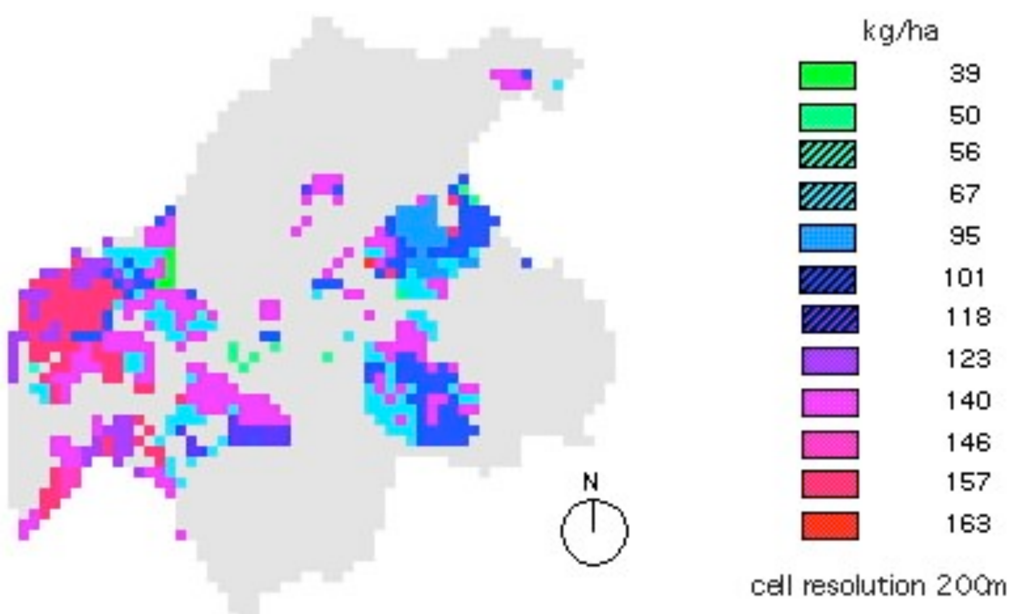


Figure E14: Estimated Nitrogen fertilizers application (Table D2) based on scenarios 1-6 Landuse, Natural Soils Groups classification, corn yield expectation and plant nutrient recommendatioin.

Table 2. Natural soil groups (NSG), yield goals and N application for Calvert County portion of Patuxent watershed

arc/info codes	NSG codes	N application (*) (kg/ha)	(lbs/a)	Corn yield(**) goal (bu/a)	number of cells	MD SCS soil types classification
2112	A1b	50	45	45	35	ReD(Rumford-Evesboro)
2113	A1c	39	35	35	32	EvE(Evesboro)
2211	B1a	140	125	125	19	ShA(Sassafras), ShC2, M1B2(Marr), M1C3, MnC2(Matapeake), MnB2
2212	B1b	118	105	105	4	WaD3(Westphalia)
2213	B1c	67	60	60	18	ErE(Eroded land, steep)
2221	B2a	157	140	140		HoB2 (Howell)
2222	B2b	95	85	85		HyD2(Howell)
2223	B2c	95	85	85		HwE2(Howell)
2511	E1	146	130	130		WoB (Woodstown)
2521	E2	140	125	125		KpB2(Keyport), BlB2(Beltsville)
2531	E3	157	140	140	12	MuA(Mattapex), MuB2
7631	F3	123	110	110	6	OtA(Otello)
7711	G1	163	145	145	1	OcB(Ochlockonee)
7721	G2	101	90	90	1	My(Mixed alluvial)
7731	G3	56	50	50	3	Tm(Tidal marsh)

References: Natural soil groups of Maryland (1973), MD Dep. of State Planning
Soil surveys by counties(1971), USDA, MD Agricultural Experiment Station
Bandel V.A., Heger E,A. MASCAP - MD's agronomic soil capability assessment program (1994) Agronomy Dep.Coop.Ext,Service, UMD, College Park
Plant nutrient recomendations based on soil tests and yield goals Agronomy MIMEO (1995), Agronomy Dep.Coop.Ext,Service, UMD, College Park

*- when corn yield was calculated, weighted avrg. was used
*- Total N recommended is 100 lbsN/a when yield goal is 100 bu/a

Calibration

When calibrating and running a model of this level of complexity and resolution, a step-wise approach is most appropriate. The HCM covers a relatively small area and could be run at a fine 200 x 200 m resolution.

We first staged a set of experiments to test the sensitivity of the hydrologic module. It has been established that there are 3 crucial parameters that control the surface water flow in the model. These were the infiltration rate, the horizontal conductivity, and the number of iterations in the hydrologic algorithms, which effectively controlled how far water could move horizontally over one day. The infiltration rate effectively controlled the height of peaks in the river water flow. The conductivity determined the amount of flow in the low period, and by changing the number of iterations we could modify the length of the peaks and the delivery rate downstream.

Calibration of the hydrologic module was conducted against the USGS data for the one gaging station on the watershed. First the model was calibrated for the 1990 data, and then it was run for 7 consecutive years (1990-1996). Figure E15 displays the annual dynamics of rainfall for 1990-1996, which shows that this period gives a good sample of various rainfall conditions that may be observed on the watershed, 1994 being the wettest year and 1991 the driest year. The results displayed in Figure E16 are in fairly good agreement with the data and may be considered as model verification, because none of the parameters were changed after the initial calibration stage for 1990.

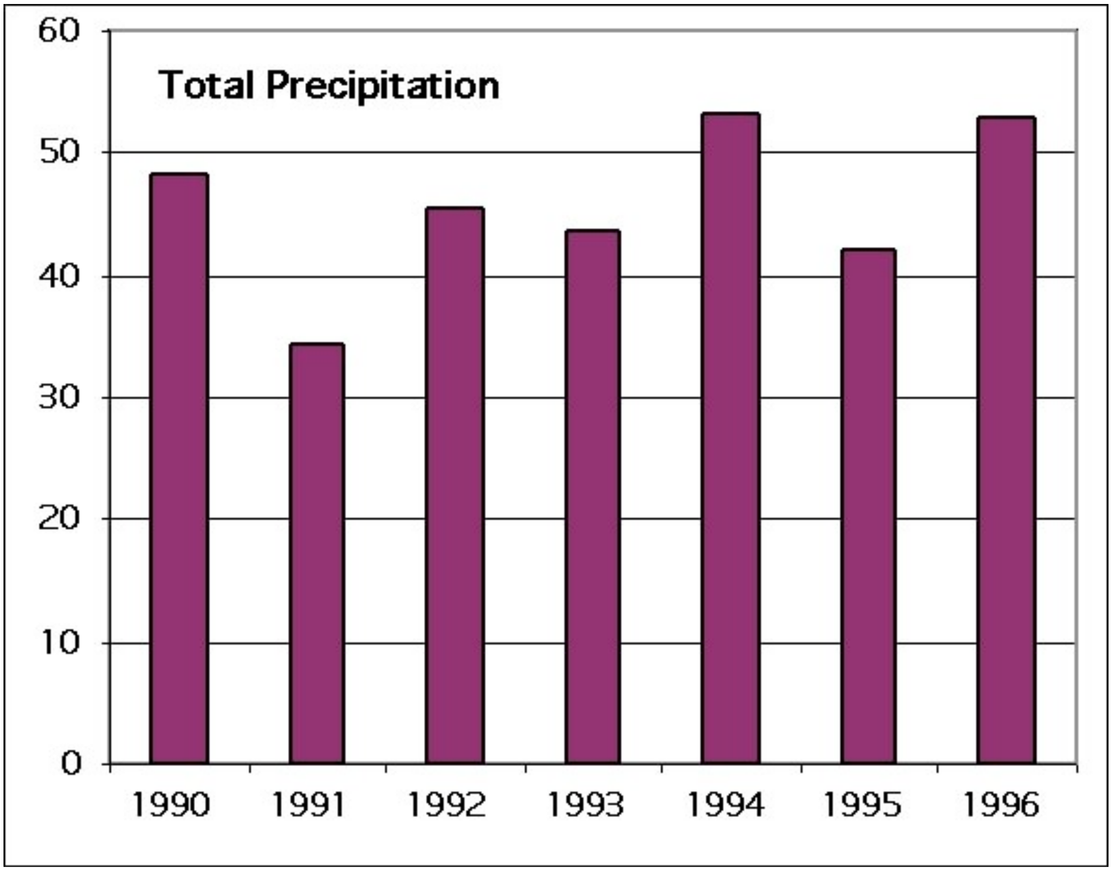


Figure E15: Annual precipitation in Hunting Creek (inches).

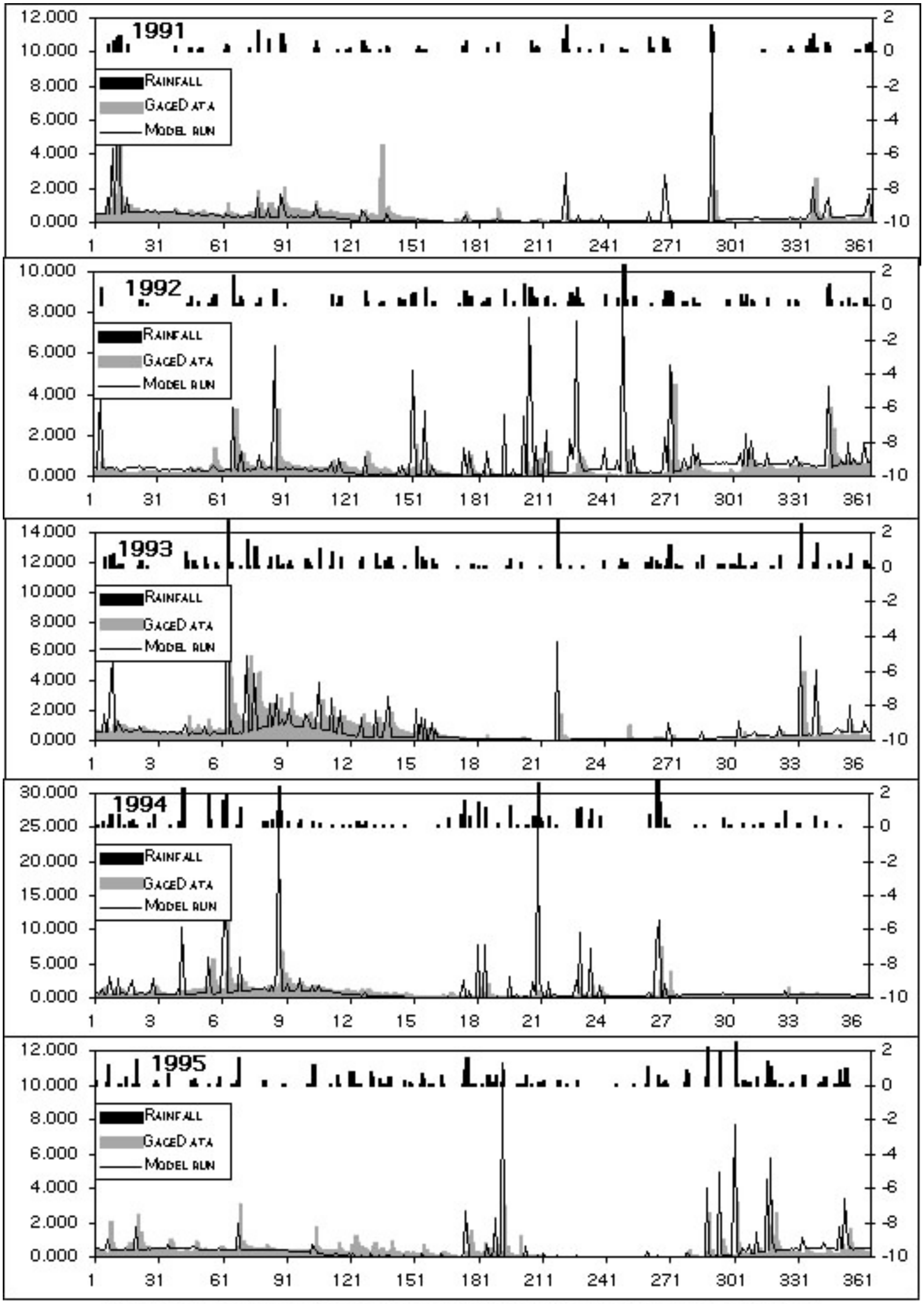


Figure E16: Calibration of the hydrologic module. Rainfall data scale on the right, flow scale on the left.

Several sources of potential error can be identified:

- Daily total precipitation is used in the model. Therefore we cannot distinguish between a downpour and a drizzle if the total amount of rainfall delivered over one day is the same. The runoff associated with these events can in fact be quite different.
- There is no climatic station located directly on the watershed. Therefore we use interpolated data from two stations nearby. However some rainfall events can be very local and therefore will not be properly simulated. The sensitivity analysis showed that the overall annual flows are highly sensitive to particular climatic time series and to the spatial patterns of climatic data.
- We also cannot exclude the chance of errors in the input data.

Nevertheless the general hydrologic trends seem to be well captured by the model. We did not have any reliable data to calibrate the spatial dynamics of ground water. However we examined the simulated total amount of water in saturated and unsaturated storage to make sure that the model is in quasi-steady state with respect to groundwater. The dynamics of these integrated values were in good agreement with the total amount of rainfall received by the watershed, responding with a lower level of the groundwater table in dry years and a rising water table during wet periods.

The comparison of flows at gaging stations is instrumental to analyze model output, calibrate and evaluate model performance. It integrates a wealth of 2-dimensional spatial information in a normalized one-dimensional fashion. For example, such spatial characteristics as infiltration rates, soil porosity, hydrologic conductivity are spatial and usually associated with a particular soil type. They define spatial flow over the landscape. Based on the elevation and link map coverages these flows are accumulated in the river network. We do not have spatial data for flow across the whole landscape, however the results observed at particular gaging stations are defined by the waterfall from all the watershed, taking into account the available spatially explicit information. Another way to view the output of a spatial model, which is especially important to localize potential accumulations of water and other spatial inconsistencies, is to output the model variables as a series of maps that can then be compiled into graphic animations. The format of a report such as this is not well suited for displaying this kind of output; further model output in map form is presented at http://giie.uvm.edu/PLM/HUNT.

Once the watershed hydrology was mimicked with sufficient accuracy, the calibration of the water quality component could be started. The nitrogen module was put into play, and the simulated nitrogen concentrations in the Hunting Creek were compared to the data observed at the USGS gaging station. It should be noted that unfortunately the station is located fairly high [does this mean upstream or vertical elevation?] on the watershed, so that it actually accounts only for a relatively small portion of the watershed. However since there is no better information available, we had to confine our calibration to this data set.

There are four major sources of nutrient loading in the watershed:

- Atmospheric deposition (data in mg/L were downloaded from the National Atmospheric Deposition Program web site (NADP, 2000)
- Discharge from sewage treatment plants (this input has been considered negligible, since in this watershed all sewage undergoes tertiary treatment (land application); however the indirect flows of nitrogen from these sources are worth further consideration in the future);
- Discharge from septic tanks (calculated as a function of discharge per individual tank multiplied by number of dwelling units multiplied by 2.9, the average number of people per dwelling unit in Maryland);
- Application of fertilizers in agricultural and residential habitats (estimated based on the

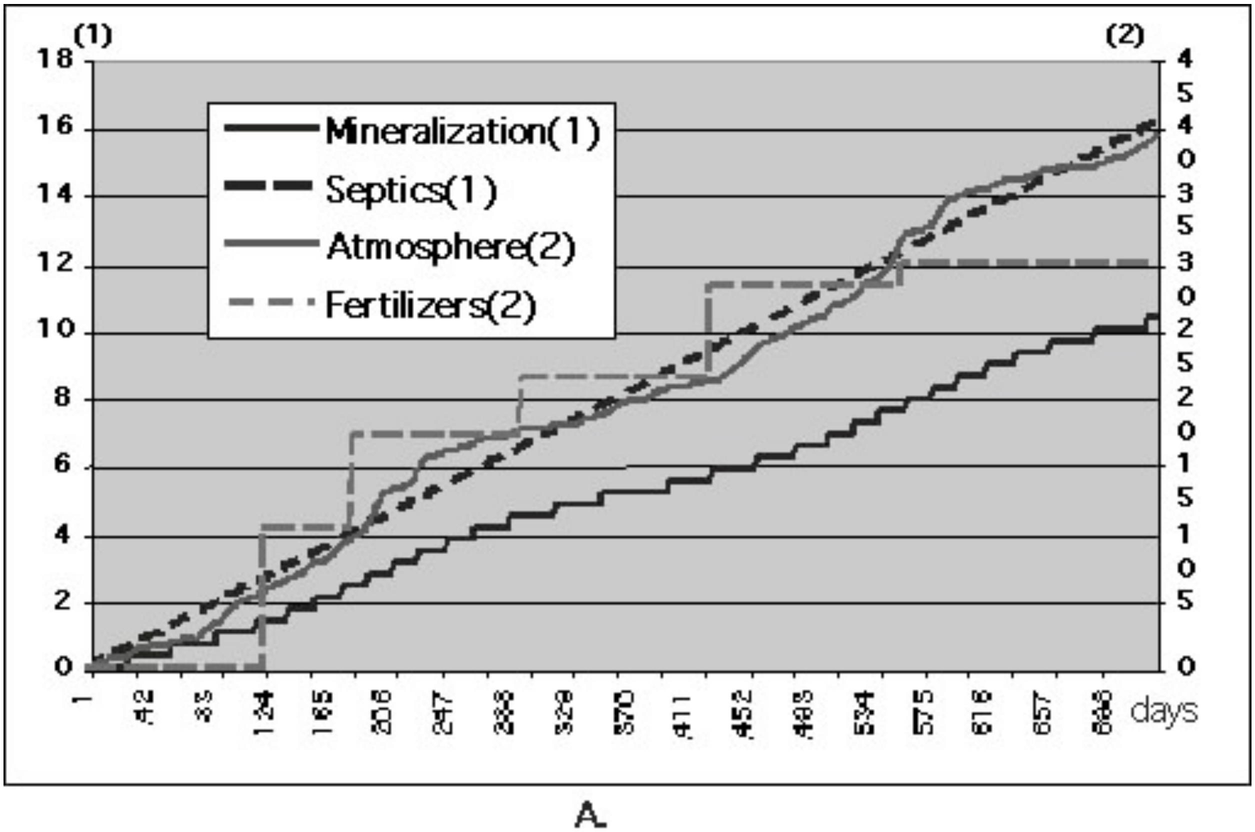
yield and soils map available from MOP);

- Mineralization of dead organic material.

The relative contribution of each of these sources is presented in Figure E17. Currently it appears that the fertilizers and the atmospheric deposition represent the major sources of nitrogen pollution on the watershed, delivering almost 80% of total nitrogen to the area. However the fate of nitrogen from different sources may be different, and one of the main uses for the simulation model is to track the pathways of nutrients from different sources to the estuary.

The model was able to reproduce the trends of nitrogen concentration at the gaging station (Figure E18). It should be noted that the water quality data are quite patchy, and a considerable time period remains unaccounted for by the observations. In addition, it may be fairly easy to miss a peak water flow while obtaining the samples, which is important because the nutrient concentrations tend to be the highest during peak flows. Therefore, the water quality data are likely to represent the baseflow concentration, and consequently they usually underestimate the true long-term nutrient dynamics.

In addition to the daily nitrogen dynamics we obtained a fairly good fit for the annual average concentration (Figure E19). This increases our confidence in the model performance, since it shows that the model does a good job of predicting the integral fluxes of nutrients over the watershed. This type of analysis is especially important when comparing the various scenarios of development in the region.



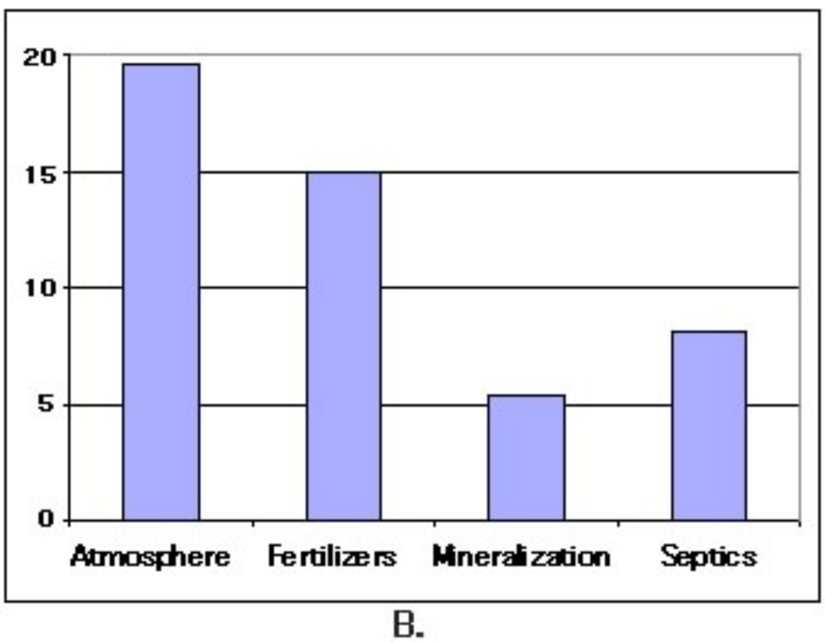


Figure E17: Nitrogen loading for the Hunting Creek watershed. A. Annual dynamics of total nitrogen loading (N kg/ha). B. Total annual nitrogen loading (N kg/ha).

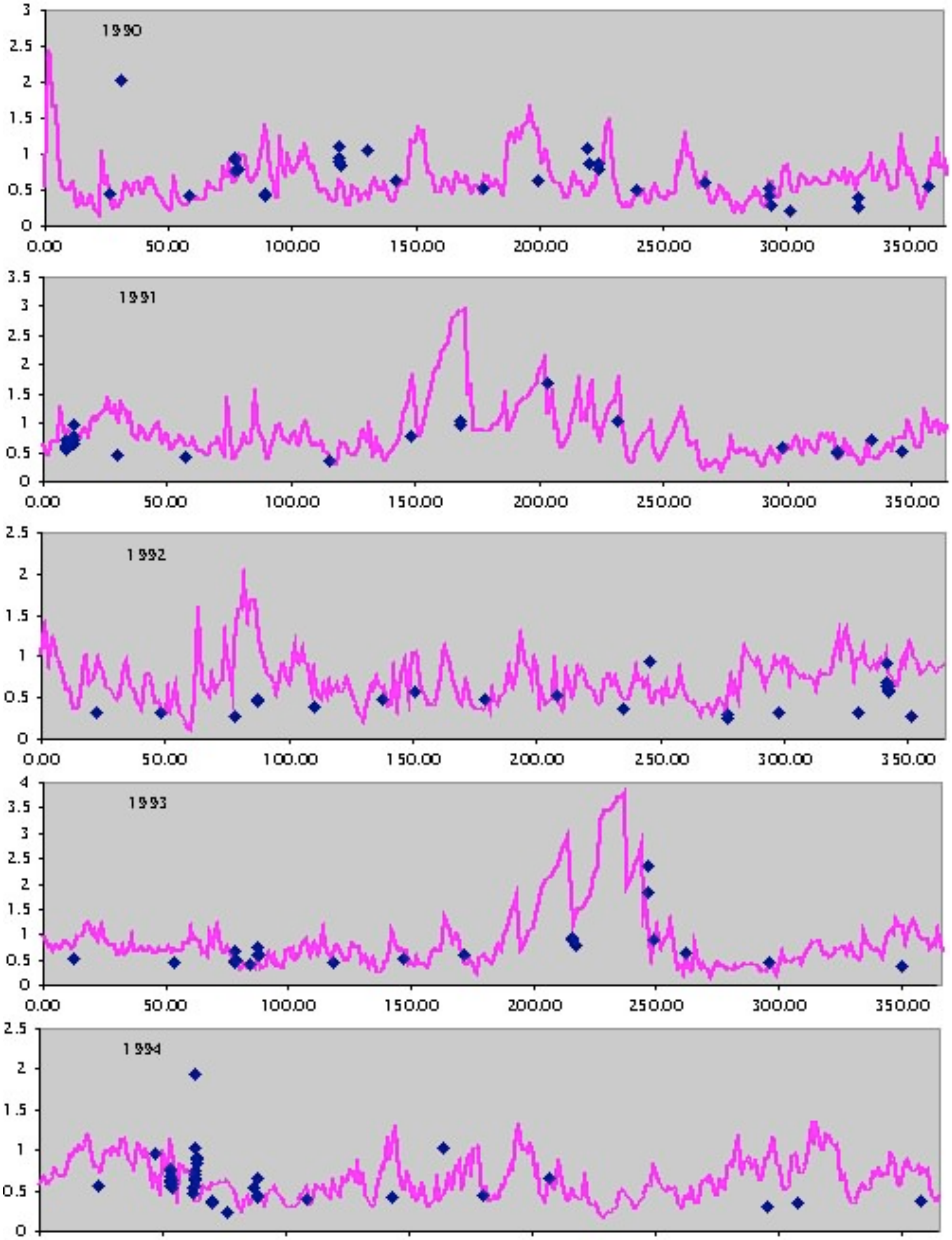


Fig.22. Calibration for total nitrogen concentration (mg N/l) in Hunting Creek (1990-1994).
◆ - data, ▬ - simulation results.

Figure E18: Calibration for total nitrogen concentraion (mg N/l) in Hunting Creek (1990-1994).

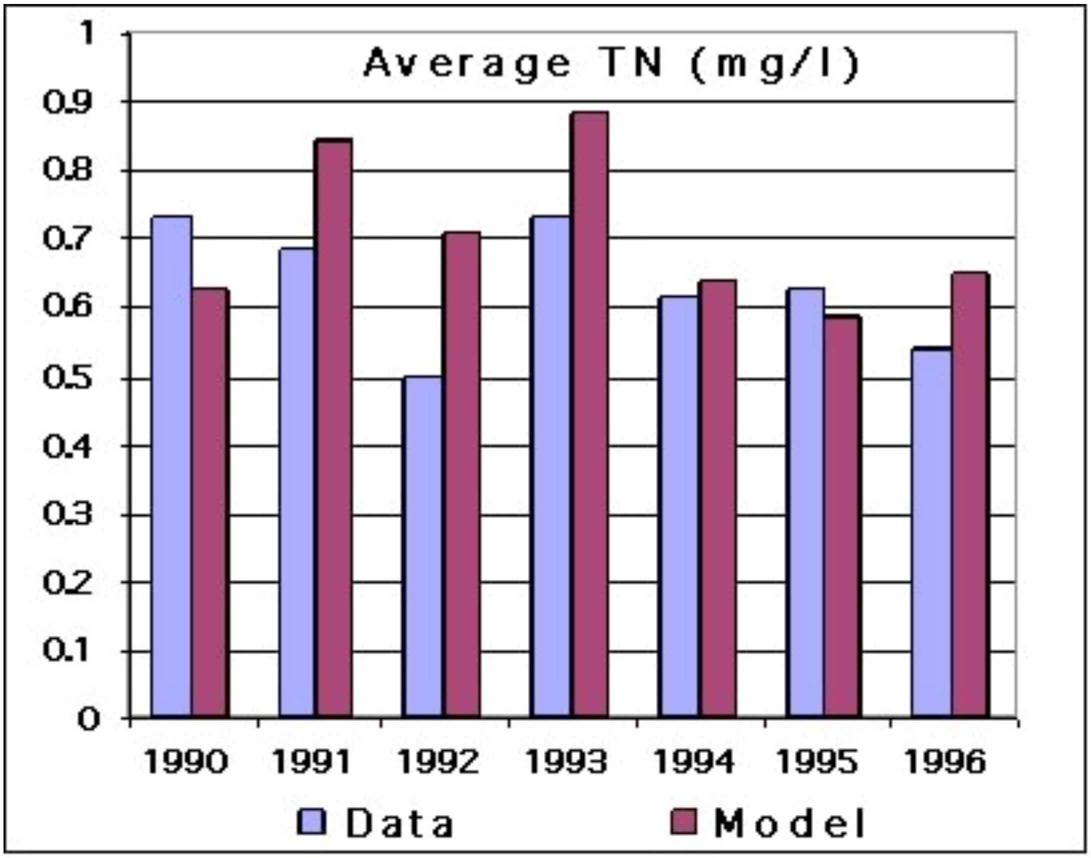


Figure E19: Comparison of average annual concentrations of total nitrogen in the model and in the USGS data.

Appendix E. Complete Hedonic Model Results

Note: For all tables, * designates significance at the 99% confidence level, ** at the 95% confidence level and * at the 90% confidence level.**

Princeton

	Value	t-value	Significance
(Intercept)	5.841795	22.628620	***
log(Liv.Area)	0.120660	2.980930	***
PropTax	0.000023	7.616288	***
log(Imp.Val)	0.492173	15.525403	***
log(LotAcres)	0.056711	7.088482	***
D2AIRPRT	-0.000003	-2.632035	***
D2TERMNL	-0.000005	-5.268763	***
D2BEACH	-0.000002	-2.059370	**
log(D2URBAN)	0.005666	3.545321	***
log(D2WATER)	0.029099	2.909319	***
D2SPARK	-0.000001	-0.452113	
D2MPARK	-0.000006	-2.670469	***
D2LPARK	-0.000004	-2.291483	**
septic	-0.033873	-1.210663	
MED.HH.INC	0.000002	6.075908	***
YRS.OLD	0.003429	4.192200	***
X2004	0.221439	4.569923	***
X2003	0.121811	4.218379	***
X2002	-0.046085	-1.864297	*

Residual standard error: 0.212356157 on 904 degrees of freedom
Multiple R-Squared: 0.86804643
F-statistic: 330.383032 on 18 and 904 degrees of freedom, the p-value is 0

New Brunswick

	Value	t-value	Significance
(Intercept)	3.862272117	7.784	***
log(Liv.Area)	0.044267533	1.978	**
log(PropTax)	0.740824135	24.658	***
log(Imp.Val)	0.060514694	2.992	***
log(LotAcres)	0.030204035	4.671	***
D2URBAN	-0.0000034	-2.788	***
D2WATER	0.000003037	1.553	
log(D2SPARK)	-0.032390188	-4.626	***
D2MPARK	0.00000328	2.073	**
log(D2LPARK)	0.018523277	3.065	***
D2BEACH	0.118599398	0.652	
log(MED.HH.INC)	-0.060697449	-1.423	
X2004	-0.00000181	-5.006	***
X2003	0.108584843	6.094	***
X2002	0.059706562	1.136	
two.story	0.084119989	4.381	***
log(D2CONTAM)	-0.038084012	-2.035	**

Residual standard error: 0.179703539 on 1636 degrees of freedom
Multiple R-Squared: 0.746015737
F-statistic: 300.333998 on 16 and 1636 degrees of freedom, the p-value is 0

Freehold

	Value	t-value	Significance
(Intercept)	6.864103484	52.017	***
log(Liv.Area)	0.153810913	10.230	***
PropTax	0.0000224	11.060	***
log(Imp.Val)	0.379310069	26.862	***
log(LotAcres)	0.076947841	14.298	***
D2AIRPRT	-0.000003805	-6.254	***
D2URBAN	0.000000522	1.442	***
URBAN	-0.000001529	-3.870	***
D2HIX	-0.054165729	-2.906	***
water100	-0.000001564	-1.717	*
CAFRA	0.12283769	1.554	
D2SPARK	-0.108707096	-5.698	***
D2MPARK	-0.000002134	-2.535	**
D2LPARK	0.000003342	2.901	***
House.age	0.000005157	8.350	***
P.VAC	-0.000484706	-2.189	**
BEACH1	0.920358857	7.075	***
BEACH2	0.228835343	3.010	***
D2UN.FOR	-0.093984101	-3.324	***
MED.HH.INC	0.000001994	9.958	***
YRS.OLD	-0.000590074	-1.488	
X2004	0.21457084	10.831	***
X2003	0.069924564	5.231	***
X2002	-0.08038887	-5.834	***

Residual standard error: 0.206935735 on 3642 degrees of freedom

Multiple R-Squared: 0.816098456

F-statistic: 702.698918 on 23 and 3642 degrees of freedom, the p-value is 0

Long Branch

	Value	t-value	Significance
(Intercept)	4.328915	41.594135	***
log(Liv.Area)	0.152283	10.003516	***
log(PropTax)	0.540172	33.870147	***
log(Imp.Val)	0.198997	18.633249	***
log(LotAcres)	0.014761	3.101666	***
log(D2URBAN)	0.001645	2.035350	**
D2WATER	-0.000005	-2.521515	**
al.siding	-0.079108	-2.605319	***
water100	0.092479	2.371969	**
D2RETAIL	0.000003	5.549714	***
D2TERMNL	0.000004	4.712920	***
ENV.SENS	0.097060	4.297184	***
D2SPARK	-0.000007	-4.481393	***
D2MPARK	0.000016	12.888312	***
D2LPARK	-0.000004	-4.910191	***
D2CLUB	-0.000014	-15.154449	***
BEACH1	0.257726	7.577603	***
BEACH2	0.045963	3.441603	***
P.BLK	-0.441276	-17.430229	***
MED.HH.INC	0.000002	11.346297	***
X2004	0.210045	2.331569	**
X2003	0.094982	6.791040	***
X2002	-0.075863	-5.265047	***
NEW	-0.023541	-2.640912	***
OLD	0.035323	6.601767	***

Residual standard error: 0.286529738 on 5991 degrees of freedom

Multiple R-Squared: 0.791114432

F-statistic: 945.407298 on 24 and 5991 degrees of freedom, the p-value is 0

Toms River

	Value	t-value	Significance
(Intercept)	6.53910765	58.918	***
log(Liv.Area)	0.12264428	10.164	***
log(PropTax)	0.48033716	48.449	***
log(Imp.Val)	0.1224917	11.604	***
log(LotAcres)	0.06662789	13.869	***
D2AIRPRT	0.00000233	4.946	***
two.story	0.04504362	7.353	***
CAFRA	0.06518192	4.920	***
D2URBAN	0.00000676	12.570	***
log(D2TERMNL)	-0.04351577	-6.378	***
log(D2WATER)	-0.02441301	-8.600	***
D2HIX	0.00000274	4.417	***
FLOOD.SFHA	0.11694168	11.533	***
water100	0.09819143	4.086	***
D2SPARK	0.00000064	0.937	
D2MPARK	-0.00000145	-2.476	**
D2LPARK	0.00000348	8.156	***
log(House.age)	-0.01731101	-3.312	***
P.VAC	0.11925327	2.208	**
NEW	-0.02124044	-2.936	***
ENV.SENS	0.03576768	2.828	***
D2UN.WET	0.00000955	3.451	***
D2UN.FOR	0.00000242	12.125	***
MED.HH.INC	-0.10721191	-5.250	***
P.OWN.OCC	-0.187534	-3.347	***
P.BLK	0.16079237	13.034	***
X2004	0.05966077	4.851	***
X2003	-0.13445492	-10.394	***
X2002	0.60186646	3.588	***
BEACH1	6.53910765	58.918	***

Residual standard error: .235852405 on 10653 degrees of freedom

Multiple R-Squared: .700414343

F-statistic: 858.831454 on 29 and 10653 degrees of freedom, the p-value is 0

Toms River fringe

	Value	t-value	Significance
(Intercept)	6.731011	7.499685	***
log(Liv.Area)	0.026877	2.249284	**
log(PropTax)	0.340076	18.545340	***
log(Imp.Val)	0.437247	21.682702	***
log(LotAcres)	0.021011	4.265779	***
D2AIRPRT	0.000013	4.205505	***
CAFRA	0.065052	2.901526	***
D2URBAN	0.000007	7.251006	***
log(D2TERMNL)	-0.123247	-4.123807	***
D2HIX	-0.000009	-8.743043	***
FLOOD.SFHA	0.139489	6.850560	***
log(D2AIRPRT)	-0.153852	-2.039607	**
D2SPARK	0.000016	9.018082	***
log(D2MPARK)	0.019294	3.529258	***
log(D2LPARK)	-0.031372	-2.982611	***
House.age	0.000779	2.317747	**
P.VAC	0.132573	1.142796	
NEW	-0.041170	-4.229298	***
D2UN.WET	0.000033	4.999717	***
MED.HH.INC	0.000001	2.772275	***
X2004	0.204360	8.701728	***
X2003	0.108891	4.610720	***
X2002	-0.081881	-3.273724	***

Residual standard error: .240302666 on 3665 degrees of freedom
Multiple R-Squared: .749568464
F-statistic: 498.624471

South Coast

	Value	t-value	Significance
(Intercept)	7.58304197	13.950	***
Liv.Area	-0.00000455	-0.379	
log(PropTax)	0.60313328	22.357	***
log(Imp.Val)	0.21887201	11.994	***
LotAcres	0.29967843	3.361	***
D2URBAN	-0.00001675	-9.090	***
log(D2WATER)	-0.04492021	-5.354	***
al.siding	-0.84312141	-2.646	***
log(D2TERMNL)	-0.27611963	-5.215	***
ENV.SENS	0.19013456	7.165	
D2SPARK	-0.00002119	-3.606	**
I(D2SPARK^2)	0	3.350	***
D2MPARK	0.00002689	7.354	***
D2LPARK	0.00002245	11.790	***
BEACH1	0.23709516	4.488	***
BEACH2	0.11549023	4.774	***
log(House.age)	-0.04620058	-3.913	***
D2UN.FOR	-2.34914171	-5.356	***
P.BLK	0.00000404	5.059	***
MED.HH.INC	0.17097883	4.593	***
X2004	0.02475577	0.670	
X2003	-0.16870714	-4.286	***
X2002	0.0289379	2.228	**
OLD	7.58304197	13.950	***

Residual standard error: .312649683 on 2224 degrees of freedom

Multiple R-Squared: .738510243

F-statistic: 273.091881 on 23 and 2224 degrees of freedom, the p-value is 0

Appendix F: Quality Assurance Plan

Summary

Valuation and value transfer

The approach to this portion of the project involves using benefits transfer methodologies to assign values to land cover types based, in some cases, on their contextual surroundings. The value estimates originate from applications of a broad range of methods and span a broad quality range, and the transfer of values from their point of origin to the target New Jersey land cover also introduces error and uncertainty. To address this, the project team maintained transparent links to the primary studies on which the estimates are based and employed a "data quality grading" system, as outlined in Costanza et al. 1992. This system can deal with the full range of data quality from statistically valid estimates to informed guesses. It assigns a numerical grade to each estimate based on assessments of the: (1) quality of models used; (2) quality of data; and (3) degree of acceptance. We implemented a simplified version of this system by creating three classes of studies, A, B, and C according to their underlying data quality (see Table 2).

GIS mapping

Since the valuation approach involves using benefits transfer methods to assign values to land cover types based, in some cases, on their contextual surroundings, one of the most important issues with GIS quality assurance is the reliability, both in terms of categorical precision and accuracy, of the land cover maps used in the benefits transfer. The team used rigorous methods to insure that the process of applying value multipliers to the maps remained error free. This involved checking area calculations to ensure that units and unit conversions are consistent, ensuring the integrity of the linkages between land cover classes and value multipliers, checking the integrity of tabular joins, and conducting manual calculations for selected records to double check certain calculations conducted in batch mode.

Hedonic analysis

This refers to the statistical disaggregation of housing prices into a schedule of marginal unobserved attribute prices and is used to empirically derive valuations for environmental amenities. Among the critical issues for hedonic analysis are the accuracy and completeness of the property sales data, accuracy of the spatial data and measurements used to derive spatial attributes, sampling strategies, rules for inclusion or exclusion of problematic observations, and analytic methods. Because of the extremely technical nature of this method, a full description of all of these is beyond the scope of this document.

Dynamic modeling

The Patuxent Landscape Model, on which this part of the study was based, has been extensively calibrated, reviewed and published (Costanza et al. 2002). The team used this model to derive relationships between spatial patterns and the provision of ecosystem services addressed in the model. The quality of these estimates can be tied to the (published) quality of the underlying model.

Data Sources

Valuation and value transfer

The data sources for this component are published studies, which have been fully referenced in the report.

GIS mapping

The two most important inputs for mapping ecosystem service values are land cover and sub-watershed boundaries (by which ecosystem service values have been summarized). Both of these have been obtained from NJDEP.

Hedonic analysis

Property sales data, including address and information on structural attributes and sales price and date, were obtained in tabular form from First American Real Estate Solutions. These records were address-geocoded, and a number of spatial attributes were derived for each observation, including control variables (e.g. distance to highway on-ramps) and variables for which values are being derived (e.g. distance to nearest park or open space). To geocode and derive these spatial attributes, a number of ancillary data sets were used (source given in parentheses). Further details on each data source is available within the metadata contained for each data layer. Layers include:

- streets/ highways (Geographic Data Technology Inc., now TeleAtlas)
- locations of downtowns/employment centers/business clusters (New Jersey Department of Community Affairs, Office of Smart Growth [NJ DCA/OSG])
- flood zones (Federal Emergency Management Agency)
- water bodies/ watercourses (New Jersey DEP)
- boundaries of public protected open space (state, county, city parks and forests, etc.; New Jersey DEP)
- Census block group boundaries (US Census Bureau)
- public transit lines and stops (GDT/TeleAtlas)
- highway exits/on-ramps (GDT/TeleAtlas)
- noxious facilities/ polluters/ major industrial sites/ Superfund sites etc./ hazardous waste sites, etc. (New Jersey DEP)
- local zoning (NJ DCA/OSG)
- school district boundaries/ school district average test scores (US Census Bureau and New Jersey Department of Education)
- shopping centers (GDT/TeleAtlas)
- Digital elevation model/slope (US Geologic Survey)

After sampling these records, a subset were analyzed using multiple regression techniques.

Dynamic modeling

Data sources for this component are detailed with the published model (Costanza et al. 2002)

Proxy measures

GIS mapping

Because ecosystem services are not mapped, the team used land cover as a proxy for ecosystem services. Using its database of valuation studies, the team was able to quantify the relationship between land cover and the ecosystem services provided for a large number of land cover types.

Hedonic analysis

As described in the main text, it was determined in the course of the hedonic analysis that the addition of school quality data to the regression model did not increase the statistical validity of the results and in some model runs actually decreased the statistical validity. It appears that the reason for this is a high degree of multicollinearity between school quality and area income. For that reason, the final model runs presented in this report exclude school quality as an independent variable, which in effect makes area income a proxy for school quality.

Historical data

None necessary for the study.

Data Comparability

Valuation and value transfer

As described earlier, the team maintained transparent links to the primary studies on which the estimates are based and also employed a "data quality grading" system, as outlined in Costanza et al. (1992). This system can deal with the full range of data quality from statistically valid estimates to informed guesses. It assigns a numerical grade to each estimate based on assessments of the quality of the underlying models, the quality of the data, and the degree of scientific acceptance of the methods. Data were coded for quality, and these codings were carried through the arithmetical calculations to help assess the quality of the results.

Hedonic analysis

A large number of value estimates for a variety of environmental resources have been derived using hedonic analysis. However, few are specific to New Jersey. This part of the study valued a set of environmental amenities specifically for New Jersey. As such, it avoided the traditional pitfalls of value transfer, where a value derived in one locale may not be truly applicable elsewhere.

GIS Data Standards

Most of the data used in the project were obtained from the New Jersey Department of Environmental Protection, or other state agencies and are presumed to meet the NJDEP spatial data standards. Some original spatial layers were created, including ecosystem service values by watershed and geocoded properties, with associated attributes. In all cases, the data processing was rigorously documented and metadata were created so as to meet the NJDEP standards. While it is expected that there are some slight spatial inaccuracies in the address geocoding of the property data, doing a full accuracy assessment of the geocoding is beyond the scope of this study because of its extremely high cost and time requirements.[28]

[28] The vendor of the particular data product used in the study, First American Real Estate Solutions, does not supply GIS data but only tabular data with addresses. Hence spatial accuracy is irrelevant from the vendor's perspective, except for errors in recording of addresses (which are difficult to assess because it would require visiting municipal offices and reviewing paper documents). Therefore, the project team address geocoded the transaction records, using the given addresses and streets data as a reference layer. The geocoding process generates a success rate, i.e., how many records were correctly geocoded and how many could not be located on a street segment. Hence, the project team can determine the percentage of records omitted, but it is very difficult to assess the accuracy of the records that were included. Unfortunately, there is little that can be done to meaningfully assess this accuracy without making expenditures that are well beyond the level of available funding for this task. To assess the accuracy of the geocoding process with the smallest degree of statistical rigor would require sampling to get representation across a wide array of geographic conditions and would be extremely expensive because the errors are not constant over space, but relate systematically to various underlying factors. For instance, geocoding mathematically interpolates the position of a given house on a street segment (i.e. block), assuming that addresses are evenly distributed along the block, which often is not the case. Hence, errors are sometimes greater for longer street segments, which tend to occur in more rural and suburban areas. In other words, a fully stratified random design would be needed to adequately assess geocoding accuracy. More importantly, assessment of geocoding accuracy would take time that would be better spent on increasing the quality of the empirical research. In the case of a hedonic analysis the gains from such an accuracy assessment simply do not justify the extremely large assessment cost. As a research method, hedonic analysis is fairly inexact in that it generally only explains about 75 to 85% of the variance in property values. Therefore, the facts that the average geocoded property location may be off by a few meters, and that perhaps 2% of the properties are off by a few dozen meters, should make little difference in the results. Moreover, to assess accuracy, the actual location of a given house must be known and determining this is very difficult without actually going in the field with an accurate GPS unit. In some cases parcel

Data Validation

Valuation and value transfer and GIS Mapping

Because ecosystem service values are not directly observable on the landscape, there is no feasible way of validating them, other than through rigorous field tests, which is beyond the scope of this study.

Hedonic Analysis

Validation of the hedonic analysis was not conducted for several reasons. First, in any regression equation, validation requires "holding aside" a validation data set that has a similar distribution of attributes to the estimation data set. Given the high price of property data and the large number of additional property observations that would have been needed, such a validation was cost-prohibitive. In other words, we had just barely enough observations to properly conduct the hedonic analysis while staying within budget. Any further parsing of the observations into a validation set would have compromised the quality of the estimation data set, which is of far greater importance. This is not a significant problem, however, as validation is rarely done for hedonic analysis. One of the reasons for this it is very difficult to generate a comparable validation data set due to the fact that many combinations of housing attributes are nearly unique. Hence, there is likely to be systematic differences in a random draw of the validation and estimation sets. Secondly, validation is not very meaningful in the case of hedonic analysis, as actual "market value," what is intended to be measured, is not directly observable, but rather is indirectly inferred from sales price. This differs from common cases where validation is used in which actual empirical measures are being validated.

Data Reduction and Reporting

Various summaries of the data were used, but NJDEP has full access to the primary data for all parts of the study. All GIS data sets have been processed and stored in a set of ArcGIS Geodatabases,with full embedded metadata and will be burned onto DVD for NJ DEP.

Sampling

Hedonic analysis

Due to its high cost, the hedonic analysis was run on a sub-sample of property transaction data for the selected study areas. Knowing that we only had budget for approximately 30,000 records, we were able to sample only a small fraction of the state. We wished to sample a relatively contained area that contained a high concentration of the natural feature types we intended to value with the hedonic analysis. The samples also needed to be contiguous, rather than dispersed around the state, so as to have sufficient statistical power to make estimates for a given housing market or neighboring housing markets, as well as to limit the amount of predictor variable data that would need to be coded. We chose to sample within Monmouth, Middlesex, and Ocean counties based on the high degree of aquatic features, parks, protected areas, wetlands, beaches, estuaries, and forests within them. Since data are sold by zip code, our initial sampling unit was zip codes. We chose to focus our analysis on Monmouth and Middlesex Counties and purchased data for all available zip codes within them (not all were available from our vendor, First American Real Estate. We also purchased somewhat less than half of the zip codes for Ocean County (08721, 08722, 08823, 08733, 08735, 08738, 08731, 08751,08752, 08753, 08755, 08757, 08759, 08527, 08533, 08701, 08723, 08724, and 08742) and a small number of zip codes bordering Middlesex or Monmouth County, in Somerset and Mercer Counties which were included because they contained important park lands (08873, 08520, 08691, 08540). We chose to sample properties in these sample zip

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maps can be used, but this would require up-to-date digital parcel layers with identifiers that link them with the property transaction data, which, from the team's experience in several states, is usually not the case.

codes with a sales price greater than $20,000, including only single family detached homes. To reach a sample of 30,000 transactions for these zip codes, we adjusted the sales date range from between January 2001 and the time at which the records were ordered (third quarter of 2004).

Analytic Methods and Statistical Tests

GIS Mapping

This part of the project involved vector geoprocessing, in which a watershed layer is unioned with a vector land use layer. Areas were then derived and summarized for each watershed (rows) by land use category (columns) using a cross-tabulation in Microsoft Access. Multipliers were then applied to each row using valuation data from the database.

Hedonic analysis

Following sampling, data were analyzed using multiple regression analysis. The appropriate functional form and model specifications were determined through analyzing goodness of fit measures and visual and quantitative analysis of residuals. Once functional form was selected, the optimal model specification was determined by using the multi-model inference approach developed by Burnham and Anderson (2002) [29], using Akaike's Information Criterion and Akaike weights (Akaike 1973; Akaike 1978) as a heuristic for selecting models that optimized the tradeoff between model fit and parsimony.

Errors and Uncertainty

Valuation and value transfer

As outlined above, the team used a data quality grading system to describe the full range of uncertainty in the results.

Hedonic Analysis

While some slight spatial inaccuracies are to be expected in the address geocoding of the property data, doing a full accuracy assessment of the geocoding is beyond the scope of this study because of its extremely high cost and time requirements. If a large number of properties are highly spatially inaccurate, this could bias the value estimates of environmental amenities. However, it is extremely unlikely that there are enough properties with consistently large enough spatial inaccuracies to cause such bias. Other errors that are common with hedonic analysis are omitted variable bias and multi-collinearity. In the former, the lack of a control variable in the model means that the observed estimated willingness-to-pay for some attribute (as represented by the coefficient) is biased because the included and omitted variables are correlated; as a result, the coefficient on the variable may be measuring the effects of both. In the latter, two independent variables in the model are highly correlated and hence the true effect of variable 1 may be accounted for in the model by variable 2. We used the multi-model inferential method (Burnham and Anderson 2002) described above in part to help weed out unnecessarily complex models that might be characterized by such correlation.

[29] Multi-model inferential procedures have been widely used for decades, using statistics such as Akaike's Information Criterion (which has been used since the early 1970s), Bayes Information Criterion, and Mallows Cp. Burnham and Anderson are among the latest authors to articulate a specific approach under this rubric, but many others have published on this general approach. A justification of this method or a bibliography of the extensive literature using this approach are beyond the scope of this QA statement but can be furnished upon request.

Performance Monitoring

GIS Mapping and Hedonic analysis

For both these tasks, the team produced detailed metadata, using New Jersey state standards, for all newly created data layers and rigorously documented the processing steps.

Documentation and Storage

Valuation and value transfer

All sources, data, and results have been documented and will be made available by NJDEP on a publicly accessible project web site.

GIS Mapping

All final GIS data have been made available to NJDEP through electronic media (e.g. FTP or CD-ROM). A large poster map will be printed as part of the final report.

Hedonic analysis

The data set of property transactions for hedonic analysis is proprietary and hence cannot be released to the public. However, all statistical results are contained in the final report and will be made available electronically as well.

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