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Accounting for Ecosystem Service Values in a Spatially Explicit Format: Value Transfer and Geographic Information Systems

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Introduction

The goods and services provided by natural ecosystems contribute to human welfare, both directly and indirectly, and therefore represent a significant, yet often uncounted, portion of the total economic value of the landscapes we live in (Wilson *et al* 2004a). While there are many ways that humans can value landscapes - economic, spiritual, and cultural - the ability to estimate the *economic* value of the ecosystem goods and services provided by them is increasingly recognized as a necessary condition for integrated environmental decision-making, sustainable business practice and land-use planning at multiple geographic scales and socio-political levels of analysis - global, national, regional and local (Bingham *et al* 1995; Millennium Assessment 2003; NRC 2005).

Ecosystem services, by definition, are the benefits people obtain either directly or indirectly from ecological systems (Daily 1997; Wilson & Carpenter 1999). They include products such as food, fuel and fiber; regulating services such as climate stabilization and flood control; and nonmaterial assets such as aesthetic views or recreational opportunities. Ecosystem goods and services occur at multiple spatial scales, from climate regulation and carbon sequestration at the global scale, to flood protection, water supply, soil formation, nutrient cycling, waste treatment and pollination at the local and regional scales (de Groot *et al* 2002; Ricketts *et al* 2004). They also span a range of degree of connection to human welfare, with those like carbon sequestration being less directly connected, while food, raw materials, and recreational opportunities are more directly connected (Farber *et al* 2002; Wilson & Carpenter 1999). Because of this connection to human welfare, environmental managers are increasingly being challenged to assess the economic values associated with ecosystem goods and services.

In this paper, we present a conceptual framework for the application of spatially explicit value transfer to assess ecosystem goods and services provided by different landscape types across multiple spatial scales. First, we briefly elucidate a formal system for classifying and valuing ecosystem goods and services associated with natural and semi-natural landscapes. Second we describe a methodology developed for conducting value transfer in a spatial context using economic data, ecological principles and Geographic Information Systems (GIS) technology. Third, we demonstrate the method by showing preliminary results from the EcoValue Project©, a web-based decision support system based at the University of Vermont that uses spatially explicit value-transfer methods. We conclude with observations on the future of spatially explicit ecosystem value transfer and its potential role in the science and management of landscapes.

Valuing Ecosystem Services

After extensive international peer review, the concept of ecosystem services has recently been adopted by the United Nations' sponsored Millennium Ecosystem Assessment (MA) program (see http://www.millenniumassessment.org). One reason is that ecosystem goods and services form a pivotal link between economic and ecological systems as well as the economists and ecologists who study them. 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 people value. Through laws, market choices and policy decisions, individuals and social groups make tradeoffs between these goods and services to maximize human values. In turn, these decisions directly affect the ecological structures and processes by engineering and construction and/or indirectly by modifying the physical, biological and chemical processes of the landscape.

Although a range of associated goods and services have been referred to in the literature (Costanza *et al* 1997; Daily 1997; de Groot *et al* 2002), the Millennium Ecosystem Assessment (2003) provides a sensible grouping of four primary categories based on functional differences.

Provisioning Goods produced or provided by ecosystems	**Regulating** Benefits obtained from regulation of ecosystem processes	**Cultural** Non-material benefits from ecosystems
• food • fresh water • fuel wood • genetic resources	• climate regulation • disease regulation • flood regulation	• spiritual • recreational • aesthetic • inspirational • educational
Supporting Services necessary for production of other ecosystem services • Soil formation • Waste Treatment and Nutrient cycling • Primary production		

Figure 1: Ecosystem Goods and Services

As this list shows, not all ecosystem goods and services are inherently substitutable with one another. For any given landscape, there are many different services that may be provided, each of which offers a unique contribution to human welfare. For example, a forested landscape may provide fuel wood or food sources, it may help regulate climate through carbon sequestration, it may prevent soil erosion and provide humus for soil formation and it may also provide aesthetic beauty and recreation opportunities. All of these goods and services contribute to the total value provided by the functioning ecological system.

Ecosystem goods and services provided by any given landscape type—forest, wetland, river—can thus potentially yield a range of values to humans. While acknowledging that human values for such ecological systems can extend from the spiritual to the utilitarian (Goulder & Kennedy 1997), the term value as it is employed in this paper has its conceptual foundation in neoclassical economic theory (Freeman 1993; Krutilla 1967). Simply put, economic value is the amount of money a person is willing to give up in order to get an ecosystem good or service (WTP), or the amount of money required to give up that good or service (WTA).

As Figure 1 suggests, ecosystem goods and services may also be divided into two broad categories: (1) the provision of direct *market* goods or services such as food, pollution disposal, and raw materials; and (2) the provision of *non-market* goods or services which include things like climate regulation, habitat for plant and animal life, and the satisfaction people derive from a nice view of a white sand beach or coral reef.

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 (Bingham *et al* 1995). When there are no explicit markets for services, more indirect means of assessing economic values must be used. A subset of economic valuation techniques commonly used to establish WTP when market values do not exist are identified below[1].

- **Avoided Cost** (AC): services allow society to avoid costs that would have been incurred in the absence of those services; flood control (barrier islands) avoids property damages, and waste treatment by wetlands avoids incurred health costs.
- **Marginal Product Estimation** (MP): Service demand is generated in a dynamic modeling environment using production function (i.e., Cobb-Douglas) to estimate value of output in response to corresponding material input.
- **Factor Income** (FI): services provide for the enhancement of incomes; water quality improvements increase commercial fisheries harvest and thus, incomes of fishermen.
- **Travel Cost** (TC): service demand may require travel, whose costs can reflect the implied value of the service; recreation areas attract distant visitors whose value placed on that area must be at least what they were willing to pay to travel to it.
- **Hedonic Pricing** (HP): service demand may be reflected in the prices people will pay for associated goods: For example, housing prices along the shore of pristine freshwater lakes tend to exceed the prices of inland homes.
- **Contingent Valuation** (CV): service demand may be elicited by posing hypothetical scenarios that involve some valuation of alternatives; people would be willing to pay for increased water quality in freshwater lakes and streams.
- **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*.

Table 1:Non-Market Valuation Techniques

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 economic value generated by a naturally functioning ecological system can be estimated using Avoided Cost (AC), can

[1] This list of non-market valuation techniques is not intended to be all-inclusive. Rather, it is intended to reveal the breadth of available empirical techniques that have been and are currently being, explored in the field of ecosystem service valuation.

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) is used for estimating property values associated with aesthetic qualities of natural ecosystems. On the other hand, Contingent Valuation (CV) surveys are often used to estimate the economic value of less tangible services like critical wildlife habitat or biodiversity. In our research, the full suite of ecosystem valuation techniques is used to account for the economic value of goods and services provided by a natural landscape.

The model of total landscape value used in this paper is based on the ecological-economic idea of functional diversity, linking different ecosystem structures and processes with the output of specific goods and services, which can then be assigned monetary values using the range of valuation techniques described above (Turner 2000). Thus, key linkages can be made between the diverse structures and processes associated with any given land cover type, the landscape and habitat features that created them and the goods and services that result (Wilson *et al* 2005). Once delineated, economic values for these goods and services can then be assessed by measuring the diverse set of human preferences for them. In economic terms, for example, the natural assets of the coastal zone can thus yield direct (fishing) and indirect (nutrient cycling) use values as well as non-use (preservation) values of the coastal system. Once accounted for, these values can then be aggregated to estimate the total value of the system (Anderson & Bishop 1986).

In sum, the concept of ecosystem goods and services is useful for three fundamental reasons. First, it helps to synthesize essential ecological and economic concepts in a dynamic conceptual system. Second, it allows us to make use of the best available ecological and economic tools to reveal meaningful values for critical ecological systems. And finally it can be used by both researchers and decision makers to transparently evaluate tradeoffs between land use change and human well being.

The Contextual Variability of Value Transfer

The growing sophistication of estimating the non-market value of ecosystem services is matched only by the rising costs of conducting individual empirical assessments for site-specific environmental changes. Unfortunately, however, only rarely can policy analysts and decision makers afford the luxury of funding, designing and implementing an original study for estimating the economic value of particular ecosystem good or services in a specific location. As a result, information from past studies published in the economic literature has been used to provide a meaningful basis for directing environmental policy and management (Desvousges *et al* 1998).

Value transfer by definition involves the adaptation of existing valuation information or data to new policy contexts with little or no data[2]. The transfer involves obtaining an estimate for the economic value of non-market goods or services through the analysis of a single study, or group of studies, that have been previously carried out to value similar goods or services. The transfer itself refers to the application of estimated point values, derived utility functions, and other information from the original 'study site' to a 'policy site'(Desvousges *et al* 1998; Loomis 1992). Value transfer has become an increasingly

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[2] Following Desvouges et. al. (1998), the term 'value transfer' is used 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 welfare functions more generally.

practical way to inform decisions when primary data collection is not feasible due to budget and time constraints, or when expected payoffs are small (EPA 2000; NRC 2005). As such, the transfer method is increasingly seen as an important tool for landscape managers and 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.

Although the transfer method is increasingly being used to inform policy decisions by public agencies, the academic debate over the validity of the method continues (Downing & Ozuna 1996; Kirchhoff *et al* 1997; Smith 1992). We accept the premise that primary valuation research will always be a "first-best" strategy for gathering information about the value of ecosystem goods and services. In other words, value transfers will always represent a compromise solution. However, when primary 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 more error prone than value transfer itself.

Thus, it is increasingly clear that with sufficient limitations and recognition of the inherent context sensitivity of value estimates, prior empirical studies can provide a basis for estimating the value of ecosystem goods and services involving sites other than the study site for which the values were originally estimated. Most importantly, as the richness, extent and detail of information about the *context* of value transfer increases, the accuracy of estimated results will likewise improve.

Here is where engagement with the concept of ecosystem goods and services and the use of tools like Geographic Information Systems (GIS) come to the foreground. Although some economists have raised awareness of the need to pay attention to the spatial and ecological characteristics of sites in relation to transfers (Bateman *et al* 2002; Eade & Moran 1996; Lovett *et al* 1997; Ruijgrok 2001), practitioners in the field have not yet effectively standardized the decomposition of transfers into spatially homogeneous units, which are widely recognized as being similar at different locations. Since ecologists have developed such classifications (i.e., land cover types), it is useful to explore whether it is possible to determine the economic values for the ecological goods and services provided by similar ecosystem types and then transfer those values from one location to another using basic ecological principles (de Groot *et al* 2002; Farber *et al* 2002). The challenge is to make value transfer spatially explicit by disaggregating complex landscapes into constituent land cover units and ecosystem service types that can be effectively transferred from one site to another.

Spatially Explicit Ecosystem Service Value Transfer

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 attributed with ecosystem services and the values they provide to people (Wilson *et al* 2004a). In simplified terms, the technique discussed here involves combining one land cover layer with another layer representing the geography to which ecosystem services are aggregated - i.e. 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. [3]

Spatial disaggregation 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 common principle in geography is that spatially aggregated measures of geographic phenomena tend to obscure local patterns of heterogeneity (Fotheringham *et al* 2000; Openshaw *et al* 1987). Analogously, aggregate measures of non-market values, while useful, can also obscure the heterogeneous nature of the underlying resources that provide those services and thus provide misleading results. For example, an aggregate measure of ecosystem services at the global level may indicate significant amounts of a land cover type associated with nutrient cycling and waste treatment, such as estuaries (Costanza *et al* 1997). This measure does not tell us, however, whether the estuaries are distributed evenly throughout the world or are all clustered in one region. Obviously, those two possibilities have significantly different ramifications for resource use and landscape management. Not only does a clustered pattern of estuaries imply that some regions have more than others, but it also means that the social cost of losing one estuarine system is much higher in the areas of scarcity than in the areas of clustering.

By mapping individual ecosystem types at higher levels of resolution, we can begin to identify areas where there is local scarcity or abundance of a given service-yielding cover type, helping us to prioritize areas of critical concern. The aggregation units used in ecosystem service mapping efforts should be driven by the intended policy or management application, keeping in mind that there are tradeoffs to reducing the aggregation unit resolution too much. For instance, a local conservation program targeted at altering land management for individual large property owners might want to use zoning parcels as aggregation units. However since such mapping would yield far too much information for state-level application, a state agency whose programs affect all lands in the state (e.g. a water resources agency) might use small watersheds as units. When using ecologically based aggregation units, like watersheds, another question is what scale to use. Because watersheds are nested, there is no clear answer as to this question. To use the wetlands example again, we may find that summarizing total area of wetlands by HUC-8[4]watersheds is sufficient for our purposes in that wetlands tends to be evenly distributed throughout them. On the other hand, we may find that in certain environments, wetlands cluster within a watershed; for instance they may tend to form in the lower reaches and less in the upper. Such a pattern could only be picked up by using finer grained watersheds. Understanding such clustering patterns may have important management implications, such as in conservation reserve design.

The first step in geoprocessing involves clipping both input layers to the same spatial extent. In some cases, the aggregation units may be nested within the extent boundary, for example when HUC 12 watersheds are used as aggregation units and the extent boundary is a HUC-6 watershed containing those sub-units. In other cases, they may be overlapping, such as where watersheds are used as aggregation units and a state boundary

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[3] The vector data model represents spatial entities with points, lines and polygons. The raster model uses grid cells to represent quantities or qualities across space.

[4] HUC refers to the nested hydrologic unit classification system (Seaber *et al* 1987). The system ranges from 2 to 16 digits, with HUC-16 watersheds being the smallest.

is used as the extent, in which case clipping of watersheds will occur. It is important to clip both inputs to the same extent, for if, for example, the land cover map stops at a state boundary and the watershed layer includes watersheds that fall partially in the state and partially outside, those watersheds will register as having a low ecosystem service value relative to area.

After clipping, the two inputs are unioned (a geoprocessing tool in which the feature geometry of two layers is combined to the full extent of both inputs) and then areas are calculated for each of the resulting “fragment” polygons. At this point, the feature geometry of the unioned layer can be discarded. All that must be kept is the attribute table of the unioned layer. The record set of this layer is fragment polygons and relevant attributes include area, land cover code and identifier of the watershed to which the fragment belongs. This is enough information to conduct a cross-tabulation of the data that will list watersheds in the rows, land cover types in the columns, and areas in the cells. This table can then be joined back to the original watershed layer. This results in an attribute table for the watershed layer enumerating area of each land cover type by watershed. This methodology involves an additional step if the land cover categorisation in the original input layer is not the same as the intended output categorisation. In the case of ecosystem service valuation, this is often the case because valuation studies often apply to broad categories, such as “forest” rather than to more precise “deciduous forest” or “coniferous forest”, which are often coded in land cover maps (Anderson *et al* 1976).

Once basic ecological units (e.g., land cover types) have been enumerated for each watershed, a total ecosystem service value for a given watershed is then calculated by multiplying the value per unit area for that ecosystem service by the area of the given cover type for that watershed. The economic values used to estimate the values associated with each ecosystem good or service are drawn from the existing non-market valuation literature. As mentioned previously, all ecosystem goods and services associated with a given spatial unit are not inherently substitutable with one another. One particular cover or land use type within a geodatabase layer may have multiple services related to it. A forest may provide fuel wood or food sources, it may help regulate climate through carbon sequestration, it may prevent soil erosion and provide humus for soil formation and it may also provide aesthetic beauty and recreation opportunities. All of these goods and services contribute to the total value provided by each functioning ecological system.

Putting it all together, the total ecosystem service value of a given cover type for a given watershed can thus be determined by adding up the individual, non-substitutable ecosystem service values associated with that cover type. The following formula is used:

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

Where A(LUi) = Area of Land Use (i)
and V(ESki) = Annual value of Ecosystem Services (k) for each Land Use (i).

In this manner, aggregate ecosystem service values for relatively homogonous landscape units can be determined by summing up all the specific ecosystem service values associated with a given unit. The results can then be divided by total landscape area at multiple scales of analysis (i.e., Huc6, Huc8, or Huc12) to give an indication of the prevalence of areas providing high ecosystem service values on the landscape. Using this approach, ecosystem service values can then be mapped and reported in graphic detail, providing decision makers with a more ecologically based view of how economic values are spread across the natural landscape.

The EcoValue Project©

Here, we briefly demonstrate the applicability of the concepts and methods reviewed above by describing an approach being developed under the auspices of the EcoValue Project currently based at the University of Vermont (Wilson *et al* 2004b). The EcoValue Project (hereafter referred to as EVP) 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 EVP provides *academic* researchers and *non-commercial* stakeholders with the ability to account for and track environmental service values in a customized, spatially explicit format. The system combines GIS and relational database technology in order to: (1) Link together available peer-reviewed economic valuation literature and ecological data in a transparent environment; (2) Allow users to interactively generate maps, graphs and economic statistics for specific parcels of land at multiple scales. The result is a multi-user platform that provides valuation data to researchers, decision-makers, and public stakeholders working in a spatially explicit mapping environment (see http://ecovalue.uvm.edu).

Currently, the EVP is being used to generate ecosystem service value estimates for the State of Maryland and the Northern Forest region. As discussed previously, the quality of the original studies used in any value transfer will ultimately determine the overall quality and scope of the final value estimates (Brouwer 2000; Desvousges *et al* 1998). Currently only the peer reviewed studies that are focused on ecological systems found in North American temperate regions are included in the EVP. This focus on is due to the consideration of their contextual similarity to the study sites in Maryland and the Northern Forest region. Using data search engines such as ISI Web of Science® and the Environmental Valuation Resource Inventory (EVRI™), the research team periodically reviews the best available economic literature and selects valuation studies which conform to the following decision rules[5]:

- Published in the peer-reviewed literature
- Limited to results that can readily be translated into spatial equivalencies—(i.e., per ha; per acre)
- Focused on regions in North America and Europe
- Focused primarily on non-consumptive resource uses

For the purpose of aggregation and comparison, all economic values in the EVP are then standardized to USD-2001 ha-1 per year. Conversion to 2001 dollar equivalents is accomplished using the Consumer Price Index (CPI) and conversion to dollar equivalencies is accomplished using available foreign exchange data. When original data is not reported in a spatial equivalent (i.e., per acre or per ha) additional information is sought from the study and augmented with information from secondary sources (i.e., GIS

[5] Current decision rules are iterative and open to change.

census data or ecological boundary data) to interpolate spatial equivalency (Woodward & Wui 2001). However, many studies in the peer-reviewed economic literature are not amenable to conversion into a unit per area measure. These studies remain in the EVP database, and are available for non-spatial queries.

Currently the EVP uses publicly available land cover/land use (LULC) codes as the primary homogeneous unit of analysis. The National Land Cover Data (NLCD 1992) is a database of satellite imagery that was collected during the early 1990's from Landsat Thematic Mapper satellites. It has been classified into 21 Land Use/ Land Cover types (LULC classes) for the United States. Resolution of this imagery (pixel size) is 30 meters. LULC information provides the fundamental link between economic values and landscape geography. Estimates for the economic value of ecosystem services are assigned to LULC types in a one-to-many relationship. For example, each LULC forest code is assigned a set of ecosystem goods and services (i.e., climate and atmospheric regulation, disturbance prevention, habitat refugium, and recreation) based on ecological functionality documented in the scientific literature (de Groot *et al* 2002). The value for these ecosystem services are then aggregated into an estimated value for each LULC type which are then associated with a particular unit of analysis (i.e., watershed). Thus, by combining the economic value estimates with land cover, the user is able to generate map images that reveal the spatial pattern of ecosystem service values across the landscape.

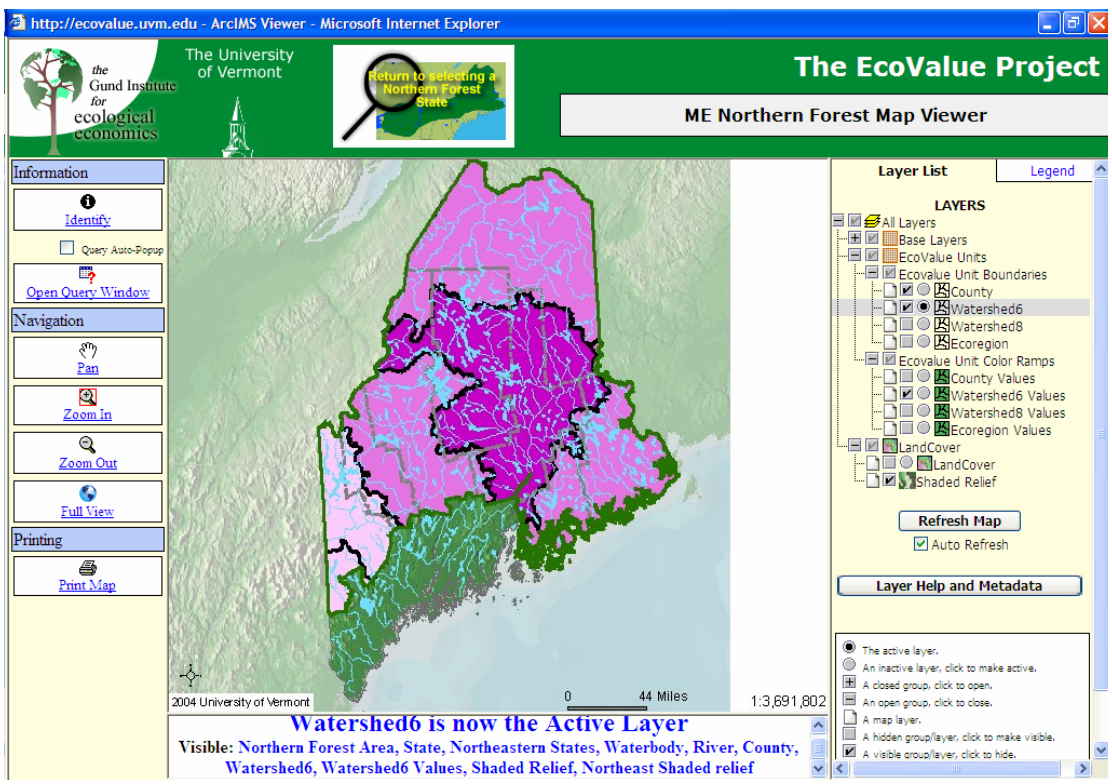

Figure 2: Northern Forest EcoValue Project map viewer with HUC 6 watershed valuation gradient active

As this screen capture of Huc 6 watershed values from the EVP shows, spatial valuation data can now dynamically be made available to users through internet browser technology. Within the EVP, spatially-explicit boundary data has been linked to the LULC and value-transfer data so that users are able to dynamically query aggregated values for at multiple spatial scales: political (state and county), hydrological (HUC 6, HUC 8, HUC 12) and ecological (Ecoregions). Although there are many types of GIS software available, the software developed by Environmental Systems Research Institute, Inc. (ESRI) is the most widely used by industries and government agencies within the United States. The ESRI software set known as ArcGIS is used extensively in the GIS component of the EVP. Data is stored within geodatabases and ArcIMS, is used as the software for delivering this data through the internet and displaying this information in the form of maps. In this system, the dynamic querying of economic values associated with these maps is made possible by using Active Server Pages (ASP). ASP uses Visual Basic scripting language (VBScript) to give users the ability to execute SQL queries of a web-based geodatabase, residing on a server at the UVM School of Business Administration, and displaying the results in real time within the user's web browser.

Future Directions

While the conceptual framework and spatial value transfer methodology described in this paper yield important and novel approaches to assessing the economic value of landscapes, such an approach should be viewed as a compliment, not a replacement, for other value transfer approaches (i.e., meta analysis, function transfer). The approach presented here represents only *one step* in what we hope will be a long process of methodological development.

There are several hurdles that must be overcome. One of the most pronounced gaps in the valuation literature is the inability to characterize the spatial and contextual variability of per unit ecosystem-service value multipliers for basic ecological units. This gap is important not just because we need to know where forests or rivers or wetlands are located within the landscape, but also because the marginal economic value of a resource is dependent on its location and the characteristics of its surroundings. Spatial context plays a role in three ways.

First, in some cases the clustering of particular ecosystem goods and services may result in "natural scale economies," such as in economic production, where the clustering together of given land cover types and their associated ecosystem services yield higher net ecosystem benefits than the same cover types or services dispersed over a large area. The analogy here is an area of a rich ore deposits clustered tightly together. Yet, while ore deposits are usually subject to extraction, ecosystem goods or services will typically be targeted for conservation or enhancement. The applicability of this postulate across landscapes will likely vary by ecosystem service, with some services being more amenable to the 'clustering' effect than others (i.e. habitat versus gas regulation).

Second, is the opposite effect. In some cases, the economic value of ecosystem goods or services derives more from scarcity than from scale economies. That is, the marginal ecosystem cost of losing a hectare of wetland in the Los Angeles Basin is likely to be far greater than the marginal cost of losing a hectare of wetland in Alaska, simply because wetlands are abundant in one and scarce in another. Hence, there is value to both spatial agglomeration and spatial dispersion of service-rendering resources. We expect the scarcity effect to be particular salient to recreational and aesthetic values. That is, the marginal social cost of losing one hectare of Central Park is likely to be far greater than that of losing one hectare of green space in a rural area with abundant green space. Currently, the valuation literature does not adequately address how non-market values vary with ecological scarcity and abundance.

Third, ecosystem service values are dependent on location relative to other thematic factors. For instance, even holding the location of a wetland relative to other wetlands, we know that not all wetlands are the same. Some wetlands may be over peaty soils, while others may be over karst-soils, influencing the macro invertebrates that might be found. Some may be surrounded by steep topography, limiting access to certain species, while others may be on flat plains facilitating access to certain species. For many species, one hectare of prime lowland is worth far more than one hectare of steep and rocky terrain. In other words, the value of a service-producing natural asset will vary with numerous other spatially varying factors.

While high resolution spatial data needed for conducting context-based ecosystem service valuation and mapping are increasingly available, a crucial limiting factor remains the availability of economic valuation studies for different ecosystem goods and services measured under different contexts. The current paucity of explicit valuation studies from different social and ecological contexts means that we must make broad generalisations when using value transfer methods to apply ecosystem value multipliers. We cannot begin to address issues of contextual variability or statistical robustness until more studies are conducted of the ecosystem service values of the same cover types in different contexts.

We encourage future researchers in the field of environmental valuation to increase reporting of contextual details about their particular study sites (i.e., spatial coordinates, ecological characteristics, socio-demographic characteristics of the study population, etc.) and to work together with ecologists to employ the evolving standard ecosystem service terminology so that value transfer research can better explain that variability of ecosystem services within and across landscapes. The ultimate goal is to have a critical mass of empirical valuation studies that will allow for comprehensive value transfers to assign value not only on the basis of land cover similarity, but also on the basis of factors like geographic scarcity or abundance, socio-demographic characteristics of the market, and spatial location of the resource.

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