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Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer

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ABSTRACT

In this paper, a decision framework designed for spatially explicit value transfer was used to estimate ecosystem service flow values and to map results for three case studies representing a diversity of spatial scales and locations: 1) Massachusetts; 2) Maury Island, Washington; and 3) three counties in California. In each case, a unique typology of land cover and aquatic resources was developed and relevant economic valuation studies were queried in order to assign estimates of ecosystem service values to each category in the typology. The result was a set of unique standardized ecosystem service value coefficients broken down by land cover class and service type for each case study. GIS analysis was then used to map the spatial distribution of each cover class at each study site. Economic values were summarized and mapped by tributary basin for Massachusetts and California and by property parcel for Maury Island. For Maury Island, changes in ecosystem service value flows were estimated under two alternative development scenarios. Drawing on lessons learned during the implementation of the case studies, the authors present some of the practical challenges that accompany spatially explicit ecosystem service value transfer. They also discuss how variability in the site characteristics and data availability for each project limits the ability to generalize a single comprehensive methodology.

1. Introduction

> Ecosystem services are the benefits people obtain either directly or indirectly from ecological systems (Millennium Ecosystem Assessment, 2003, page *v*.)

The process of identifying and quantifying ecosystem services is increasingly recognized as a valuable tool for the efficient allocation of environmental resources (Heal et al., 2005; Millennium Ecosystem Assessment, 2003). By estimating and accounting for the economic value of ecosystem services, social costs or benefits that otherwise would remain hidden can potentially be revealed and vital information that might otherwise remain outside of the economic decision making calculus at local, national, and international scales can be internalized (Millennium Ecosystem Assessment, 2005). However, achieving such an objective requires considerably better understanding of ecosystem services and the landscapes that provide them.

In this paper, we present a framework for the spatial analysis of ecosystem service values (ESVs), illustrated through three case studies. Thanks to the increased ease of using Geographic Information Systems (GIS) and the public availability of high quality land cover data sets, bio-geographic entities such as forests, wetlands and beaches can now more easily be attributed with the ecosystem services they deliver on the ground (Bateman et al., 1999; Eade and Moran, 1996; Kreuter et

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al., 2001; Wilson et al., 2004). This approach compliments the other transfer techniques discussed in this Special Issue of *Ecological Economics*.

The ability to integrate biophysical and ecosystem service valuation data is a relatively new phenomenon (Kreuter et al., 2001; Wilson and Troy, 2005). Rather than argue for a single unified methodological approach that can apply to all possible circumstances, our goal is to outline a set of decision rules that have served as the basis of our efforts in three case studies.

This paper first briefly reviews previous efforts to classify and place economic values on ecosystem services associated with natural and semi-natural landscapes in a spatially explicit manner. Second, it describes our decision making framework for conducting spatially explicit value transfer by linking analyses of non-market economic valuation data and biophysical data. Third, it describes how this framework was applied to three case studies. Fourth, it discusses limitations, including the potential variability of each implementation. The paper concludes with observations on current trends and expected future directions in spatially explicit ecosystem value transfer.

2. Spatially explicit value transfer

Value transfer involves the adaptation of existing valuation information to new policy contexts where valuation data is absent or limited.[1] For ESVs this involves searching the literature for valuation studies on ecosystem services associated with ecological resource types present at the policy site. Value estimates are then transferred from the original study site to the policy site (Desvousges et al., 1998; Loomis, 1992). Value transfer has become an increasingly practical way to inform decisions when primary data collection is not feasible due to budget and time constraints, or when expected payoffs to original research are small (Environmental Protection Agency, 2000). As such, the transfer method is now seen as an important tool for environmental policy makers since it can be used to relatively quickly estimate the economic values associated with a particular landscape for less time and expense than a new primary study (see Iovanna and Griffiths, 2006-this issue).

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 (see Wilson and Hoehn, 2006-this issue). Primary valuation research will always be a “first-best” strategy for gathering information about the value of ecosystem goods and services. However, when conducting primary research is not feasible, value transfer represents a meaningful “second-best” strategy and starting point for the evaluation of environmental management and policy alternatives. While value transfer is far from perfect, we believe that it is better than the status quo approach of assigning a value of zero to ecosystem services.

[1] Following Desvousges et al. (1998), we adopt the term ‘value transfer’ instead of the more commonly used term ‘benefit transfer’ to reflect the fact that our approach 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.

One of the biggest potential pitfalls in value transfer occurs when values are drawn from study sites that are situated in very different contexts than targeted policy sites. For example, to simply assume that the economic value of a freshwater wetland in one ecological region is going to be the same for a freshwater wetland in a wholly different region would be inappropriate. Given this, we anticipate that as the richness, extent, and detail of information about the context of value transfer increases, the accuracy of estimated results will improve. The better we are able to match the biophysical and socio-economic context of the source with the target, the more accurate our estimates will be.

While much attention has focused on the economic theory and practice of environmental value transfer itself, much less attention has been paid to the inherently spatial nature of many environmental values. As Eade and Moran (1996) note:

> The spatial dimension to economic valuation has barely been investigated…The adoption of a spatial approach to economic valuation is desirable in terms of producing more accurate economic valuation figures, for use as a repository for benefits estimates, examining spatial sustainability, and facilitating the introduction of natural capital concepts into environmental decision-making processes (p. 109).

Spatial disaggregation of ecosystem services allows us to visualize the pattern and distribution of ecologically important landscape elements and overlay them with other relevant themes (Bateman et al., 1999; Eade and Moran, 1996). 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. 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). Yet, this global measure does not tell us whether the estuaries are distributed evenly throughout the study region or are all clustered in one region-conditions that have very different implications for land use management.

To date, the number of published analyses using a spatial value transfer framework is limited. Among those studies is one by Kreuter et al. (2001), who attempted to quantify the impact of urban sprawl on the delivery of ecosystem services using LANDSAT imagery and global ecosystem service value coefficients derived from Costanza et al. (1997). The authors used satellite imagery and remote sensing software to determine the area of six land use classes in each of three watersheds in Bexar County, Texas. These estimates were then incorporated into an economic valuation model that used biome-level, global approximations of ecosystem service values (Costanza et al., 1997). Based on this analysis, the authors determined that there was a 65% decrease of rangeland and 29% increase in the area of urbanized land use between 1976 and 1991 with a resulting net 4% decline of annual ecosystem service values for that same time period. This relatively small decline was attributed to the effect of a 403% increase in the area of woodlands which were assigned the highest ecosystem service value coefficient.

While economists have certainly raised awareness of the importance of considering spatial and ecological context of sites in conducting value transfer (Bateman et al., 2002; Eade and Moran, 1996; Lovett et al., 1997; Ruijgrok, 2001), functionally meaningful classifications of ecological resources have yet to be developed for the purposes of value transfer. Ecologists have developed such classifications for characterizing the ecological function of landscapes, but these characterizations may not always be appropriate for economic applications. Our challenge is therefore to link economic valuation data to landscapes using typological characterizations that are functionally meaningful.

3. A decision framework for mapping ecosystem service values

The approach presented in this paper forms the foundation of the Natural Assets Information System™, a decision support system framework developed by Spatial Informatics Group, LLC (http://www.sig-gis.com). The framework, which builds upon the value transfer methodology, is implemented in three case studies and consists of five core steps: 1) spatial designation of the study extent; 2) establishment of a land cover[2] typology whose classes predict significant differences in the flow and value of ecosystem services; 3) meta-analysis of peer-reviewed valuation literature to link per unit area coefficients to available cover types; 4) mapping land cover and associated ecosystem service flows; 5) calculation of total ESV and breakdown by cover class; 6) tabulation and summary of ESVs by relevant management geographies and; 7) scenario or historic change analysis. These steps are described generally in the following paragraphs and in more detail in the Applications section below. Note that we limit our discussion here to the calculation of ecosystem service value flows. However, ecosystem service stocks may also be calculated through estimating the net present value of the future flow of ecosystem services.

3.1. *Step 1: study area definition*

Study area definition is an essential but often underappreciated first step, since small boundary adjustments can have large impacts on final ESV estimates. While the client's desired target area for study may correspond neatly with administrative or political boundaries those may or may not correspond with relevant bio-geophysical boundaries. For instance, the coastal boundary of a state could be defined to include areas within the statutory boundary of a state, the territorial waters of the state (3 nautical miles from the coastline), the area between shoreline and the off-shore ocean shelf, or it could even be based on characteristics of coastal aquatic features (i.e., bathymetry). Each of these boundary definitions will have a significant impact on the final results when estimating the economic value of ecosystem services delivered by the coastal zone.

3.2. *Step 2: typology development*

The development of a land cover typology starts with a preliminary survey of available GIS data at the site to determine the basic land cover types present. This is followed by a preliminary review of economic studies (see step 3) to determine whether ecosystem service value coefficients have been documented for these cover types in a relatively similar context.[3] Once this is done the GIS and valuation analysts search for required spatial data layers and valuation studies to fill in identified gaps. For instance, an initial assessment of land cover might find that forests, pastures, open water and wetlands are present in a particular study area. A preliminary review of published valuation studies may also suggest that transferable estimates exist for all of these and that, moreover, several studies break down the forest valuations into early and late successional stage. The GIS technician would then look for spatial data on forest successional stage within the study area or, lacking that, some secondary data source that could be used to model it.

3.3. *Step 3: literature search and analysis*

The collected empirical studies, preferably from a similar context, are read and analyzed to extract valuation coefficients for ecosystem services associated with each cover class in the typology. The information includes the ecosystem service and cover type valued, valuation method, year of study, and per hectare value estimates, among other attributes. If not enough applicable studies are available, additional studies may have to be located, and the results summarized. Where no studies exist in the literature but valuation estimates are essential, new empirical studies may need to be commissioned. Increasing the number of economic studies used in a value transfer project achieves several purposes. First, it fills in gaps where a particular service associated with a particular land cover may previously have been unknown. Second, multiple studies for a given ecosystem service provide a range of estimates that allow the analyst to determine if any given estimate appears unreasonable. There are three broad categories of valuation studies that exist in the field today:

- Peer-reviewed journal articles, books and book chapters, proceedings and technical reports that use conventional environmental economic valuation techniques and are restricted to an analysis of social and economic values. These are the most desirable studies.
- Non peer-reviewed publications that include PhD dissertations, technical reports and proceedings, as well as public raw data.

[2] Here, the term "land cover" incorporates aspects of both land use and land cover. Moreover, the typologies incorporate both terrestrial and aquatic resources. However, because our typologies primarily refer to cover rather than use, and terrestrial rather than aquatic resources, and lacking an adequately succinct alternative term, we use "land cover" throughout.

[3] Steps 2 and 3 generally co-occur together in an iterative fashion because the availability of valuation studies will necessarily impact the development of the land cover typology and the availability of land cover data will necessarily impact the valuation studies chosen for transfer.

- Secondary analysis (e.g., meta-analysis) of peer reviewed and/or non peer-reviewed studies that use conventional or non-conventional valuation methods.

3.4. Step 4: mapping

Map creation involves GIS overlay analysis and geoprocessing to combine input layers from diverse sources to derive the final land cover map. This is often complicated by differences in parent scale, year of creation, accuracy, data model, and minimum mapping unit for each input layer. Therefore the methods used in this step are highly variable by project and will be subject to the judgment of the GIS analyst. Often that analyst's job will include the process of conflation, through which multiple layers are combined to extract the highest quality or most relevant elements from each. For instance, one layer may depict highly precise land cover, but be somewhat out of date. A new layer may show areas of recent change, but have poor resolution. If the areas of change are small enough in extent, the two can be combined to create an improved layer. Alternately, a high quality land cover layer may lack an important category needed for the typology, such as old growth forests. Through conflation, the analyst would update mapping units designated as forest in the land cover layer with data from a stand age thematic layer.

3.5. Step 5: total value calculation

Once each mapping unit is assigned a cover type, it can then be assigned a value multiplier from the economic literature, allowing ecosystem service values to be summed and cross-tabulated by service and land cover type. The total ecosystem service value flow of a given cover type is then calculated by adding up the individual, non-substitutable ecosystem service values associated with that cover type and multiplying by area as given below.

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

Where $A(LU_i)$ = area of land use/cover type (i)and $V(ES_{ki})$ = annual value per unit area for ecosystem service type (k) generated by land use/cover type (i).

3.6. Step 6: geographic summaries

In the fifth step land cover areas and ESVs are summarized by a geographical aggregation unit. While ESVs can be mapped by the original minimum mapping unit (e.g. a land cover pixel), for large map extents with small minimum mapping units (pixels are frequently 30 m on a side or smaller), patterns may be visually imperceptible and so geographic aggregation is often warranted. Moreover, managers may be interested in visually displaying the value of ecosystem services by some geographical unit with management significance, such as town, county, or watershed.

3.7. Step 7: scenario analysis

Finally, scenario or historic change analysis can be conducted by changing the inputs in steps 4 and 5. For future scenario analysis this involves changing the land cover input to reflect a proposed management alternative and for historic change analysis it involves quantifying and valuing land cover changes in the past.

4. Case study sites

Here, the approach outlined above is examined at three different spatial scales in the United States: 1) the Commonwealth of Massachusetts; 2) Maury Island, a small island located in Puget Sound, Washington; and 3) three counties in the State of California. The first analysis was conducted by the authors under contract from the Massachusetts Audubon Society, the second, in conjunction with Northern Economics, Inc and Herrera Environmental Consulting, was conducted for King County, WA and the third, in conjunction with TSS Consultants, was conducted for the California Office of the US Bureau of Land Management.

The Massachusetts project was conducted for the Massachusetts Audubon Society's "Losing Ground" project, a technical report targeting state and local officials that described the extent and pattern of habitat loss in the state due to urbanization and land use change (see Wilson and Troy, 2003). For that project, the study site consisted of the entire state of Massachusetts, including all inland water bodies, to the coastal boundary, excluding any off-shore or nearshore areas. The final area covered an area of 2,096,042 ha.

The Maury Island value transfer study was part of a larger project conducted by Herrera Environmental Consultants, Inc. Northern Economics, Inc., and SIG for the government of King County, WA. Its purpose was to inventory and estimate the biological resources, socio-economic characteristics and ecosystem service values of a small island located in the Puget Sound, near Seattle, WA (Herrera Environmental Consultants et al., 2004). The project also sought to estimate how ecosystem service values might change under alternative development scenarios. The study area boundary (Fig. 1, next section) contained the entire land area of the island seaward to the waterward limit of the nearshore zone, which is defined as extending to the lower edge of the photic zone, or the aquatic area extending to a depth of roughly15 m (approximately 55 ft) below mean low water level. This resulted in coverage of about 2459 ha.

The California study was part of a project conducted by TSS Consultants and SIG for the California office of the US Bureau of Land Management (BLM). The purpose of the study was to assess the value of both market and non-market ecosystem service assets that could be lost in the event of a catastrophic landscape-scale fire in communities participating in BLM's wildfire mitigation programs (TSS Consultants and Spatial Informatics Group LLC, 2005). The team sampled three counties with diverse landscapes: Humboldt, Napa, and San Bernardino. Within each county analysis was limited to zip codes containing communities participating in BLM's community fire mitigation programs, resulting in 897,568, 196,650 and 2,983,224 ha respectively in each. In the cases of Napa and Humboldt counties, the study zip codes occupied over 90% of the counties' land areas, while in San Bernardino County, those areas represented a little over half of the county.

5. Applications

For all three case studies, we estimated ecosystem service value flows, broke them down by land cover type, mapped their

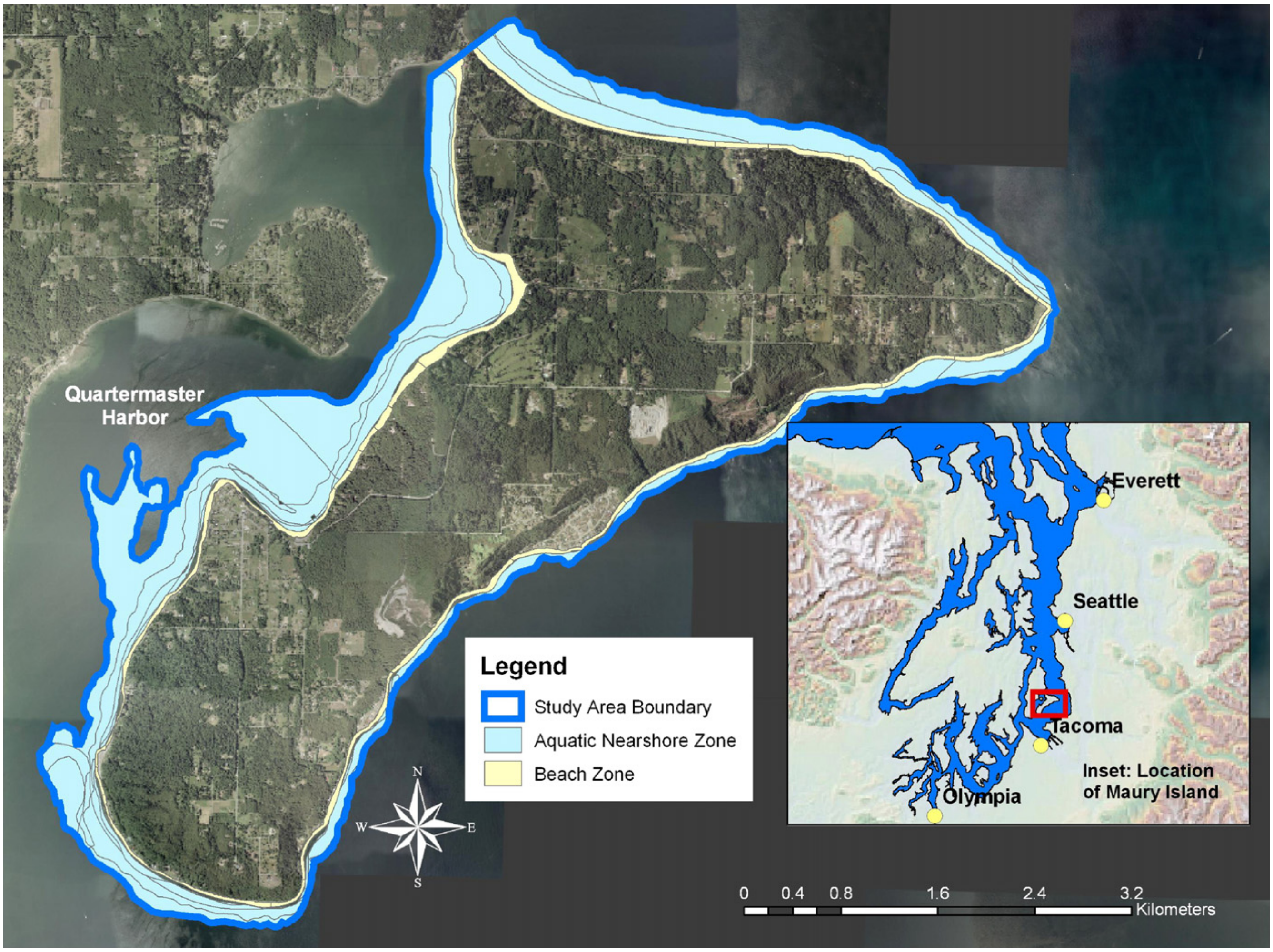

Fig. 1 – Map of Maury Island study area.

distribution, and summarized them geographically. For Maury Island we also conducted scenario analyses. Considerable flexibility was needed when applying the framework because of differences in data availability and management objectives, as well as the different needs of the clients and stakesholders.

Study area boundaries were determined in direct consultation with the clients during the earliest phases of all three projects presented here. On the one hand, in the case of Massachusetts a combination of manmade and natural boundaries was selected to define the study area so that it best captured the impacts of urbanization and land use change in the state. The inland study area border followed the official state boundary, while seaward borders followed those defined in Massachusetts' official hydrologic basins map. In California, on the other hand, only the study area zip code boundaries, nested within county boundaries, were used to define the study area. This resulted in differing levels of inclusion of estuaries and coastal embayments-features that could by definition fall either in or out of a land-based study area. For example, both Napa and Humboldt counties have embayments, seasonally flooded marshes, or estuaries at their edges, but the official Napa County boundary contains a higher proportion of those waters than Humboldt, likely because Napa fronts on an inland water body–San Pablo Bay–while Humboldt County fronts on the Pacific Ocean.

Because of the finer spatial scale and increased importance of nearshore resources in the case of Maury Island, a far more precise study area boundary definition was needed. In this case, the project team based the study extent on the edge of the photic zone surrounding the island, defined as the bathymetric limit at which the underwater floor receives light. After consulting with biologists familiar with Puget Sound, this depth was determined by the study team to be 15 m below the Mean Lower Low Water (MLLW) mark for the area around Maury Island (Fig. 1).[4] The photic zone is significant because it represents an area of high biological productivity, yielding plant communities like eelgrass, which are in turn spawning habitat for many economically valuable species.

The process of developing case-specific land cover typologies and maps differed based on site characteristics, data

[4] A complication in the boundary definition existed, however, in that the west side of Maury Island (Quartermaster Harbor) is a shallow bay that is nowhere greater than 15 m in depth. Hence, using high resolution bathymetric data, we defined the western boundary as everything east of the deepest bathymetric contour in Quartermaster Harbor.

availability and management objectives. In each case study, typologies were developed using the process discussed above. All three typologies are given in Table 1. Sample images of land cover for the three study areas, all at 1:20,000 scale, are also given in Fig. 2. These images illustrate the significant differences in the minimum mapping unit type, boundary precision and resolution for each application.

The earliest typology developed by the authors was for Massachusetts (Wilson and Troy, 2003). This case benefited from a high resolution vector land use map that previously had been created by the Massachusetts Geographic Information System. This map was based on 1:25,000 air photos, had a one acre minimum mapping unit, 21 categories and classified the state for 1985 and 1999 (we analyzed land use change between the two time periods, but only 1999 results are discussed here). Reviewing those classes in relation to a preliminary list of valuation studies suggested that those 21 categories could be combined into nine for the purposes of the project (eight classes with positive value plus one aggregated class for all non-valued or valueless types). The only typological addition we made was urban green space, which consisted of the categories "urban open space" and "participation recreation sites".

Once the typology was set, empirical valuation studies were analyzed and entered into the NaturalAssets Information System™ system and standardized to 2001 dollar equivalents.[5] This process yielded 42 viable peer-reviewed empirical studies and 65 valuation data points that were used in the final analysis. In the interests of space a complete bibliography of valuation studies is not given for any of the case studies here; rather, bibliographies are contained in each individual project report (see Wilson and Troy, 2003 for Massachusetts). Studies were filtered for inclusion not only based on land cover type, but also on contextual similarity of the study site to the 'policy site' in Massachusetts and the type of valuation method used. For example, even though many land cover types in Massachusetts support pollinators and benefit from pollination services, the only available empirical research from the lite-rature on this service used the replacement cost method (Southwick and Southwick, 1992), and the client did not want to incorporate studies that used that method (see also Heal et al., 2005). As a result of these limitations, the study yielded conservative lower bound ecosystem service value estimates for Massachusetts.

The eleven class typology for Maury Island given in Table 1 was developed by the authors in consultation with the client and other members of the consulting team (Herrera Environmental Consultants et al., 2004). Because the site was so small and accurate spatial data could be created relatively easily through digitizing, spatial data availability was less of a constraint in developing the typology than availability of valuation studies. Eventually, we found studies on all cover classes in our initially proposed typology. The valuation literature analysis yielded 43 applicable studies and a data set containing 71 marginal per hectare value estimates for numerous ecosystem services associate with the eleven cover classes, standardized to 2001 dollar equivalents.

[5] All dollar values are standardized using Consumer Price Index tables published by the U.S. Department of Labor. http://www.bls.gov/cpi/home.htm.

Table 1 – Land cover typologies for three case studies

Maury Island	Massachusetts	California
Disturbed[a]	Disturbed[a]	Disturbed[a]
Saltwater wetland	Saltwater wetland	Saltwater wetland
Freshwater wetland	Freshwater wetland	Freshwater wetland
Nearshore habitat[b]	Freshwater or coastal embayments	Estuaries
Coastal open water		Open fresh water
Grassland/herbaceous	Pasture	
	Cropland	Agriculture[c]
		Vineyards[d]
Stream buffers		Forested river buffers
Coastal riparian		
	Urban green space	Urban green space
	Woody perennial	
Beach		
Beach near dwelling		
Forest	Forest	Hardwood forest
		Conifer forest
		Mixed forest
		Second growth redwood forest
		Old growth redwood forest
		Northern spotted owl forest habitat

[a] Includes urban, barren, and unvalued land cover types.
[b] Includes intertidal salt estuaries, estuarine intertidal aquatic beds, stream mouths, sea cucumber habitat, geoduck habitat, and herring and salmon spawning grounds. All other nearshore areas were classified as "nearshore-open salt water", which had no established valuation.
[c] Includes row and hay crops and pasture.
[d] Only for Napa County.

Because the Maury Island project included such a small study area, it was essential that the spatial data be at an appropriate scale. The base land cover layer created by King County, WA classified 2001 LANDSAT data into 30 meter raster pixels. Because 30 m pixels were too coarse for the needed level of analysis, we augmented this base layer with finer resolution data where available, including 1 meter resolution impervious surface data, classified from IKONOS satellite imagery by King County, which was used to update the "urban and barren" class, resulting in differing levels of spatial precision by category.

Several of the Maury Island cover classes required digitizing or geoprocessing to derive. For instance, the beach cover class was digitized from an aerial photo. Since the valuation database differentiated beach values based on proximity to residences, manual digitization was used to further subdivide beach polygons into those proximate (defined as roughly 200 ft from the nearest house) and not proximate to residential structures. Additional categories requiring ancillary data layers and processing were 50 foot[6] stream buffers created around vector stream center lines, wetlands from the National Wetlands Inventory, a coastal buffer defined 200 ft inland from the Mean Higher High Water (MHHW) mark, and the aquatic nearshore zone, defined as the edge of the MHHW mark to a depth of 15 m below MLLW. All of these unique layers were combined into a

[6] The 50 foot distance was chosen by biologists with our partner Herrera Environmental Consulting who were familiar with the island. Buffer widths are uniform throughout the island.

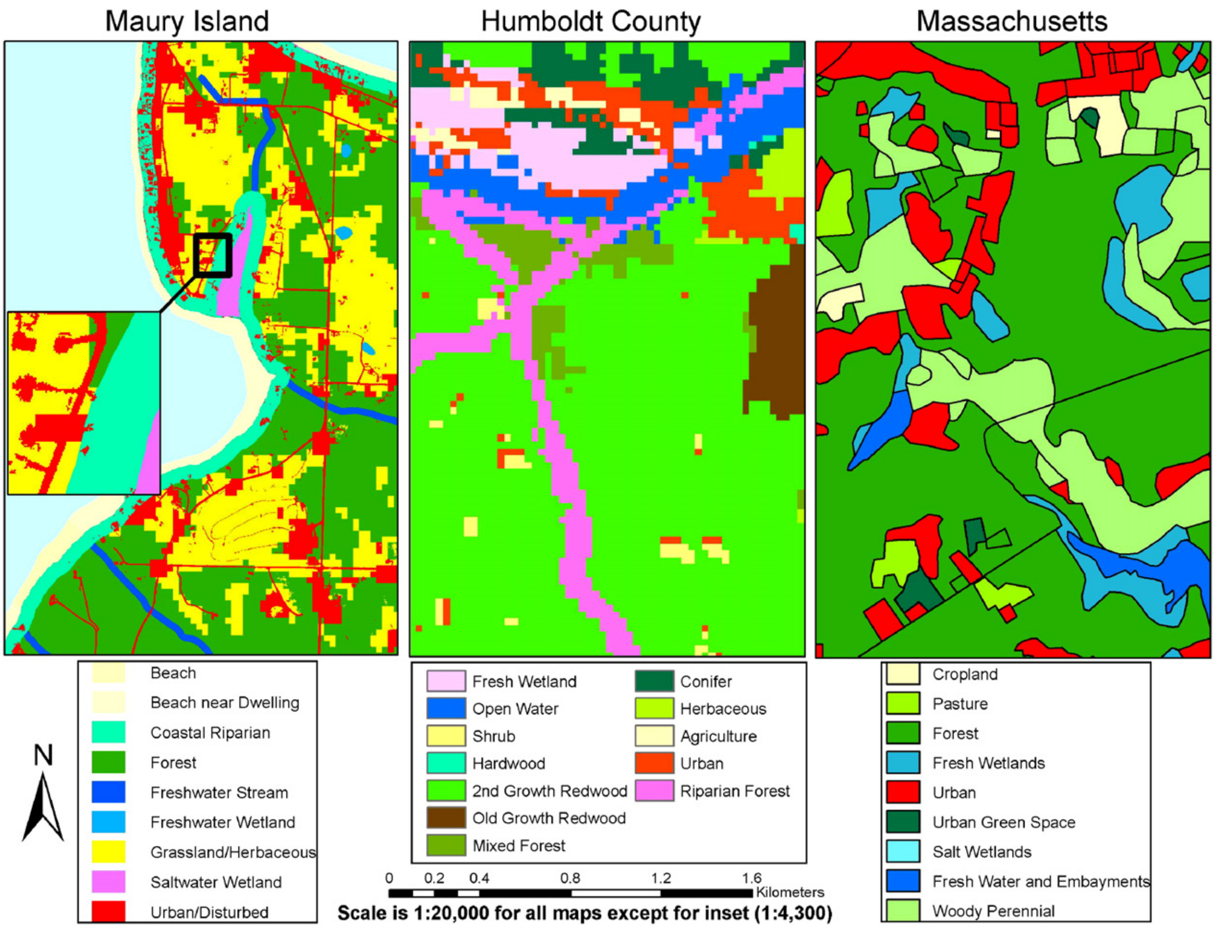

Fig. 2 – Land cover map comparison of Maury Island, Massachusetts, and Humbolt County at 1:20,000 scale.

single layer where each polygon was assigned a mutually exclusive land or aquatic type. Where the nearshore zone and beaches coincided, polygons were classified as beach, since this category had higher values and since beaches were more accurately mapped through hand digitizing.

The California case study had the most complex land cover typology (TSS Consultants and Spatial Informatics Group LLC, 2005), which was informed by the management objectives of wildfire hazard mitigation. Because forests are the cover type most subject to catastrophic wildfire, we focused on dividing forests into as many sub-categories as possible. Following discussions with BLM managers, we searched both the valuation literature and available GIS data to determine which relevant cover classes were both valued and adequately spatially attributed. This resulted in a typology of fourteen cover classes providing thirteen documented ecosystem services. Several very important cover classes, such as desert shrub and arid woodland ecosystems, which covered several million hectares in San Bernardino County, were combined into the fifteenth "unvalued" category because no appropriately transferable studies were found.[7] The resulting value transfer exercise utilized 84 empirical valuation studies, yielding a total of 205 individual value estimates of per hectare value coefficients for the fourteen valued cover types, with results standardized to 2004 U.S. dollar equivalents. Estimates were coded by time of study, location, and valuation method.

Land cover varied significantly between the counties and data sets were not always consistently available. Hence we relied on several spatially comprehensive but lower quality data sets, and augmented them where we could. The base layer used for all counties was the 2003 California Land Cover Mapping and Monitoring Program Vegetation Map (known as Calveg), a raster layer with 30 meter resolution. This was updated with data from the National Wetlands Inventory (where available) for salt and fresh wetlands and estuaries, the National Hydrography Dataset for open water and for the streams used to define 50 meter[8] riparian forest buffers, California Department of Forestry's (CDF) Coastal Redwood Vegetation layer for secondary and old growth redwood stands, and the US Forest Service's Northern Spotted Owl Dataset for spotted owl habitat. For the portion of the study areas where it was available, the US Geological Survey's 2001 National Land Cover Database (NLCD) impervious layer was

[7] Much of what is mapped as scrub in San Bernardino County is chaparral, a vegetative community which, although highly prone to fire, provides a number of significant ecosystem services whose economic values have not been well quantified.

[8] The 50 meter buffer width was chosen based on the Report of the Scientific Review Panel on California Forest Practice Rules and Salmonid Habitat (Ligon et al., 1999) which recommends a 150 foot (~ 46 m) riparian forest buffer around class 1 streams in order to preserve ecological function. Buffer widths are uniform throughout the study area.

Table 2 – Ecosystem service values by cover type for Massachusetts

Land cover type	Average $/ha/yr	Lower bound	Upper bound	Area (ha)	Total ESV flow
Cropland	$ 3427	$ 3427	$ 3427	90,087	$ 308,729,091
Pasture	$ 3412	$ 3412	$ 3412	36,940	$ 126,038,356
Forest	$ 2430	$ 2429	$ 1005	1,200,303	$ 2,916,735,578
Freshwater wetland	$ 8474	$ 38,167	$ 18,979	46,460	$ 393,701,543
Salt wetland	$ 31,084	$ 31,071	$ 24,678	8439	$ 573,164,281
Urban green space	$ 3430	$ 8471	$ 6649	58,535	$ 7,141,321
Woody perennial	$ 122	$ 122	$ 122	17,372	$ 2,119,421
Fresh water bodies/ coastal embayments	$ 38,183	$ 2427	$ 159	69,657	$ 2,659,701,737
Disturbed and urban	$ –			556,075	$ –
Total				2,093,868	$ 6,987,331,328

used to update urban areas and a combination of the 2001 NLCD tree canopy layer and the US Census's urbanized areas layer were used to designate urban green space. In Humboldt County, where a large amount of clear cutting operations occur, CDF's Cause of Landuse Change Dataset was used to update areas that had been recently clearcut.

Unlike the previous two case studies, which used predominantly vector data, the California study was completed in a raster environment. Calveg was used as the base raster layer. Information from other layers was integrated through "conditional" raster queries. In this method the user specifies a condition, which can be composed of multiple criteria. For pixels where the condition is true, it returns a constant or an array of values from another designated layer and where false it returns a different constant or values from a different layer. For instance, to create the riparian forests category, the function identifies all pixels that fall within a 50 m raster stream buffer and that are defined as a forest type in the Calveg layer, returning a new identifier for all those pixels. All pixels that fail to meet the condition retain their original values from the Calveg layer.

Once ESVs were mapped, they were aggregated to summary geographies with management significance. In the case of Maury Island, these units were property parcels, while for Massachusetts and California they were hydrologic units (tributary basins or watersheds).

Finally, for Maury Island, the King County government requested an analysis of changes in ESV flows and stocks under two alternative development scenarios: 1) enlargement of a gravel mine and an associated dock and; 2) buildout to full allowable residential zoning on the island over the course of 20 years. The former scenario was quantified by setting to zero the ESV flows of 68 ha in the proposed footprint of the 95 hectare property, as well as for the proposed footprint of the expanded dock. The latter was quantified by using a digital zoning map, flagging all those parcels that could be further built up or subdivided under allowable zoning, and simulating the loss in natural cover types in those parcels. For example, a 30-acre undeveloped parcel currently zoned as R-10 could be legally subdivided into three ten-acres parcels with one dwelling unit per parcel. To simulate this, the average impervious surface ratios associated with currently built out parcels within the R-10 zone was applied to the undeveloped parcel. In this paper, we simply detail the change in service flows in 2004 dollars for each scenario, assuming the changes were immediate. An assessment of the change in the net present value of the stock of ecosystem services, taking into account the expected gradual reduction in service flows over time in both scenarios, was conducted by Northern Economics, Inc and SIG and is detailed in the report by Herrera Environmental Consultants et al. (2004).

6. Results

Standardized ecosystem service value flows are presented below for Massachusetts in Table 2, for Maury Island in Table 3 and for California in Table 4. The California and Maury Island tables give results in adjusted 2004 dollars, while the Massachusetts results are in 2001 dollars. These tabulations show an estimated ESV flow of $2.98 billion for the California sample counties, $22.6 million for Maury Island, and $6.98 billion for Massachusetts.

The tables break down ESV flows, summed across all service types, by land cover. They also give the area in hectares and the average dollar value per hectare per year for each cover type which when multiplied together give the total ESV flow. Massachusetts results show that forests, fresh water bodies, and coastal embayments yield by far the highest ESV flow of any class, accounting for almost $5.6 billion of the total. Maury Island results show that nearshore aquatic habitat and beaches located near structures provide the highest proportion of ESVs, resulting in almost $17 million between them. The California results show that in San Bernardino County, freshwater wetlands account for the majority of the ESV flows, followed by riparian forest. While desert scrub and woodlands were valued at zero due to lack of studies, their areas are still given in this table to show how much of an impact they could have on ESV estimates if valued.

Table 3 – Ecosystem service values by cover type for Maury Island

Land cover	Ave. $/ha/yr	Lower bound	Upper bound	Area (ha)	Total ESV flow
Disturbed and urban	$ –	$ –	$ –	253	$ –
Beach	$ 88,204	$ 77,016	$ 99,391	27	$ 2,371,006
Beach near dwelling	$ 117,254	$ 140,505	$ 94,004	65	$ 7,575,825
Coastal riparian	$ 9396	$ 5542	$ 13,248	132	$ 1,244,665
Forest	$ 1826	$ 511	$ 3142	1044	$ 1,906,410
Freshwater stream	$ 1595	$ 1,231	$ 939	41	$ 66,059
Freshwater wetland	$ 72,787	$ 32,947	$ 96,095	4	$ 269,089
Grassland/ herbaceous	$ 118	$ 118	$ 118	321	$ 37,833
Nearshore aquatic habitat	$ 16,283	$ 4630	$ 27,935	565	$ 9,204,633
Saltwater wetland	$ 1413	$ 854	$ 1972	7	$ 9527
Total				2460	$ 22,685,047

Table 4 – Ecosystem service values by cover type and county for California

Description	Ave. $/ha/yr	Humboldt County		Napa County		San Bernardino County	
		Area (ha)	Total ESV flow	Area	Total ESV flow	Area	Total ESV flow
Agriculture	$ 2192	15,937	$ 34,932,508	11,210	$ 24,571,316	29,041	$ 3,657,272
Conifer forest	$ 821	114,244	$ 93,823,306	7012	$ 5,758,593	135,033	$ 10,896,564
Desert shrub	NA	0	0	0	0	4,123,497	NA
Desert woodland	NA	0	0	0	0	245,288	NA
Estuary	$ 5898	2	$ 10,085	451	$ 2,661,834	0	0
Fresh wetland	$ 10,973	9593	$ 105,261,803	1785	$ 19,592,412	74,968	$822,650,494
Hardwood oak woodland	$ 439	112,182	$ 49,293,301	59,030	$ 25,938,010	19,404	$ 8,526,125
Herbaceous	NA	83,079	0	26,769	$ 0	22,595	NA
Mixed forest	$ 826	261,920	$ 216,293,687	5511	$ 4,551,190	34,790	$ 28,729,641
Spotted owl habitat	$ 998	89,670	$ 89,487,414	0	0	0	0
Riparian forest	$ 8792	49,472	$ 434,960,966	7073	$ 62,189,858	37,854	$ 332,816,821
Redwood 2nd growth	$ 815	99,632	$ 81,185,900	511	$ 416,315	0	0
Redwood old growth	$ 950	39,661	$ 37,682,967	0	0	0	0
Shrubs	NA	22,483	NA	48,549	NA	195,273	NA
Saltwater wetland	$ 6044	549	$ 3,317,256	1396	$ 8,438,390	0	0
Disturbed and urban	0	17,379	0	7471	0	267,097	0
Urban green	$ 5605	3255	$ 18,242,491	731	$ 4,099,948	62	$ 344,531
Vineyards	$ 2192	0	0	14,178	$ 31,075,280	0	0
Open fresh water	$ 7237	7145	$ 51,707,928	12,107	$ 87,621,444	17,044	$ 123,347,887
County totals		926,202	$1,216,199,612	203,786	$ 276,914,591	5,201,946	$1,490,969,335
Grand total for all counties	$2,984,083,539						

NA = value is expected to be greater than zero but is not known.

Additional results are discussed in greater detail in Wilson and Troy (2003), Herrera Environmental Consultants et al. (2004) and TSS Consultants and Spatial Informatics Group (2005).

Table 5 cross tabulates ESVs by land cover and service type for Maury Island. The number of blank cells where positive numbers would be expected illustrates that significant gaps exist in the valuation literature. That is, not all land cover types have been valued for all possible associated ecosystem services. This results from the limited availability of economic valuation data.

For each project, maps were created to illustrate the spatial distribution of ecosystem service flows. Examples of watershed-level summary maps of ecosystem service values are given for Humboldt County in Fig. 3 (using total ESV per watershed) and for Massachusetts in Fig. 4 (using average ESV per hectare by watershed).

Taken together, these maps show the heterogeneity in the spatial distribution of resources providing ecosystem services. In Massachusetts, for instance, by far the highest per hectare ESVs occur along the coast, especially around important estuaries and bays where wetlands are prevalent. In Maury Island (not shown here), the highest values are found in beachside properties, which benefit from the extremely high amenity

Table 5 – Ecosystem service values by land cover and service type for Maury Island

Land cover	Aesthetic and amenity	Climate and atmospheric regulation	Disturbance prevention	Food and raw materials	Habitat refugium	Recreation	Soil retention and formation	Waste assimilation	Water regulation and supply
Beach	$ –	$ –	$ –	$ –	$ –	$2,371,006	$ –	$ –	$ –
Beach near dwelling	$4,442,228	$ –	$ –	$ –	$ –	$ –	$3,133,597	$ –	$ –
Coastal riparian	$ 224,009	$ –	$ 48,622	$ –	$ 509,067	$ 10,732	$ 107,842	$ 29,872	$314,520
Forest	$ 7703	$1,391,576	$ –	$ –	$ 10,041	$ 483,395	$ –	$ –	$ 13,695
Freshwater stream	$ 25	$ –	$ –	$ –	$ 24,641	$ 17,585	$ –	$ –	$ 23,807
Freshwater wetland	$ 17,866	$ –	$ 56,893	$ –	$ 85,466	$ 4203	$ –	$104,642	$ 20
Grassland/ herbaceous	$ –	$ 2649	$ –	$ –	$ –	$ 755	$ 379	$ 32,915	$ 1135
Nearshore habitat	$ –	$ –	$ –	$2,080,557	$3,518,838	$3,605,238	$ –	$ –	$ –
Saltwater wetland	$ –	$ –	$ 3770	$ –	$ –	$ 173	$ –	$ 1,474	$ 4110
Column total	$ 4,691,832	$1,394,224	$ 109,284	$2,080,557	$4,148,054	$6,493,088	$ 3,241,818	$168,903	$357,286

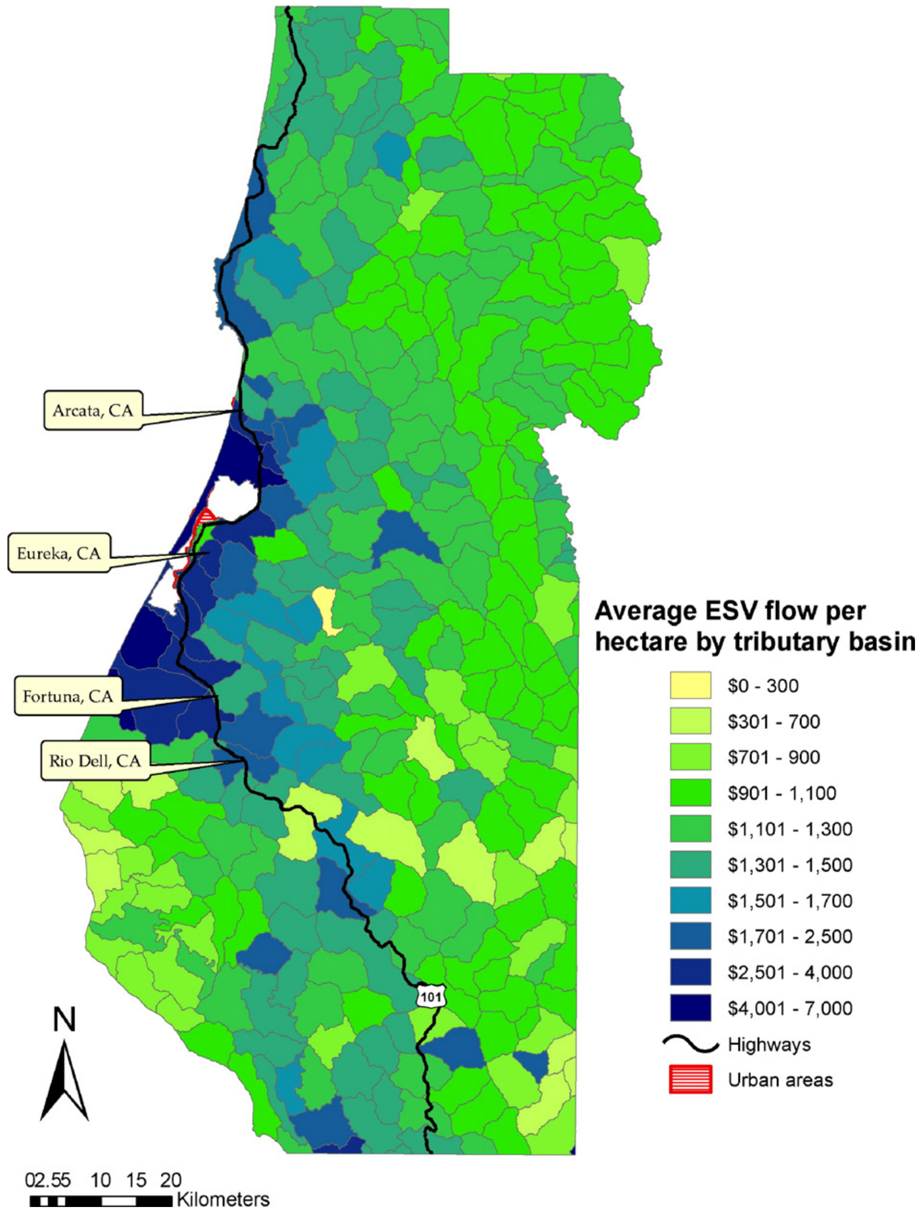

Fig. 3 – Average yearly ecosystem service value flows per hectare by Tributary Basin for Humboldt County, CA in 2004 dollars.

values and biophysical services associated with that resource. In the case of Humboldt County we see high ESVs in watersheds both near the coast and inland, especially in areas of old growth redwood forests.

The Maury Island scenario analyses estimated that under the full mine development scenario, if all mine development occurred in the first year, there would be a loss of $703,000 in yearly ecosystem service value flow in the subsequent year. Under the allowable zoning buildout scenario, if all development were to occur at once, there would be a reduction in $548,000 in the yearly flow starting in the subsequent year. Net present value of stocks under the various scenarios is given in Herrera Environmental Consultants et al. (2004). ESV's were summarized by parcel for Maury Island under current conditions and future conditions under allowable zoning buildout. The resulting percentage loss in ESV was mapped by parcel (Fig. 5). This map shows that the large inland parcels would undergo the greatest percentage reduction in ESVs under full buildout since they are currently on average the least developed.

7. Discussion: limitations and lessons learned

While this paper offers a framework for the spatial analysis of ESV's, the case studies described above clearly illustrate how each application of the framework is subject to variability relating to limitations in the available spatial data and economic valuation studies, as well as differences between site characteristics, spatial and temporal scale, and management objectives.

The availability of empirical economic valuation studies is one of the most significant constraints to spatially explicit value transfer today. As shown in Table 5, a large number of important land cover types currently have no economic valuation studies

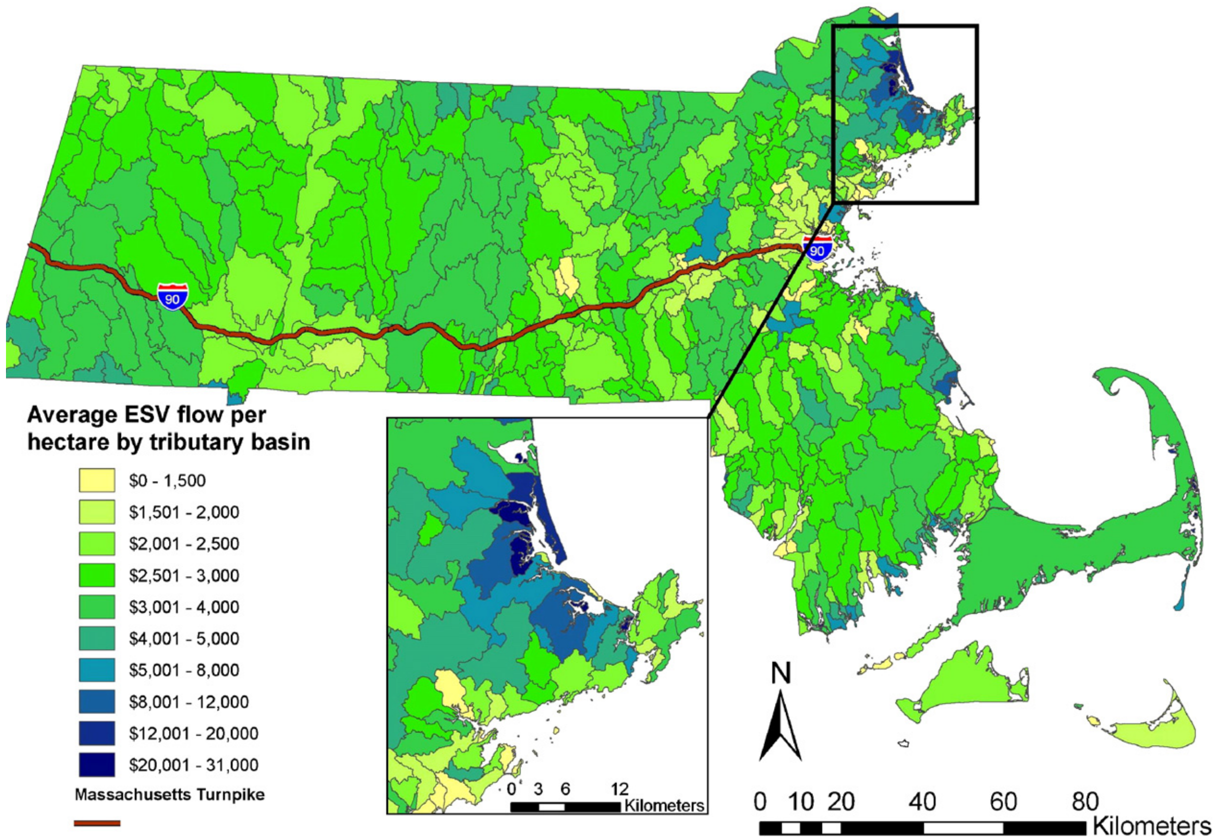

Fig. 4 – Average yearly ecosystem service value flows per hectare by Tributary Basin for Massachusetts in 2001 dollars.

associated with them and for those that do exist, generally a limited number of ecosystem services have been valued. Furthermore, as land cover is split into a greater number of more precise classes (e.g. from "forest" to "early", "middle" and "late" successional stage forest), the number of blank cells will, by definition increase.

The availability of valuation data is further limited by the fact that only economic studies whose valuation coefficients were derived in a similar context to the policy site should be used for value transfer (Desvousges et al., 1998). Yet, defining contextual similarity itself can be challenging and, because of the limited number of studies available, must involve trade-offs among specificity, reliability and applicability. This lack of comparability among studies, stemming from differences in the characteristics and context of the resources being valued, has been cited as a significant limitation in meta-analysis and value transfer (Van den Bergh and Button, 1997; Woodward and Wui, 2001). Woodward and Wui (2001) note that if enough data existed, these differences could be controlled for, but the lack of sufficient data means that biases will often result.

Three critical factors must be considered when assessing comparability between the source data and policy context. First, one must consider the biogeophysical similarity of the policy site and the study site. For instance, an economic value coefficient determined for a tropical forest cover type should not generally be transferred to temperate regions, since tropical forests and temperate forests are very different in both form and function. Second, the human population characteristics of source data must be considered. For instance, estimated values of wetlands based on water regulation or flood avoidance services provided to large downstream population centers should not necessarily be transferred to contexts with no downstream population centers. Moreover, because willingness to pay, the basis of many valuation studies, reflects preferences weighted by income, differences in the incomes of the "served population" should be approached with caution, although alternative methods, such as adjusting willingness to pay to purchasing power parity could be used in some cases. Further complicating this is the fact that the size and shape of the area of influence of an ecosystem service will likely be different depending on the service type. For water-related services it will likely be defined by hydrologic connectivity within a watershed, while for recreation and amenities it may be defined by driving distance and for gas regulation it can be the entire globe.

Third, similarity in the level of scarcity of the service should be considered. Systems with an abundance of a given land cover type are more likely to have redundancy in the services it provides. Where there is a scarcity of a natural resource type that provides a given type of service for which no substitute exists, even a small marginal loss of that resource could be devastating and, thus, the value would increase accordingly. For instance, the marginal ecosystem cost of losing a single hectare of coastal wetland in Florida Everglades is likely to be relatively low compared to the cost of losing a hectare of the Ballona wetlands in metropolitan Los Angeles, which are among the last remaining coastal wetlands in the area and which provide critical services in filtering and regulating nutrients and toxins in stormwater before entering the Santa Monica Bay (Tsihrintzis et al., 1996).

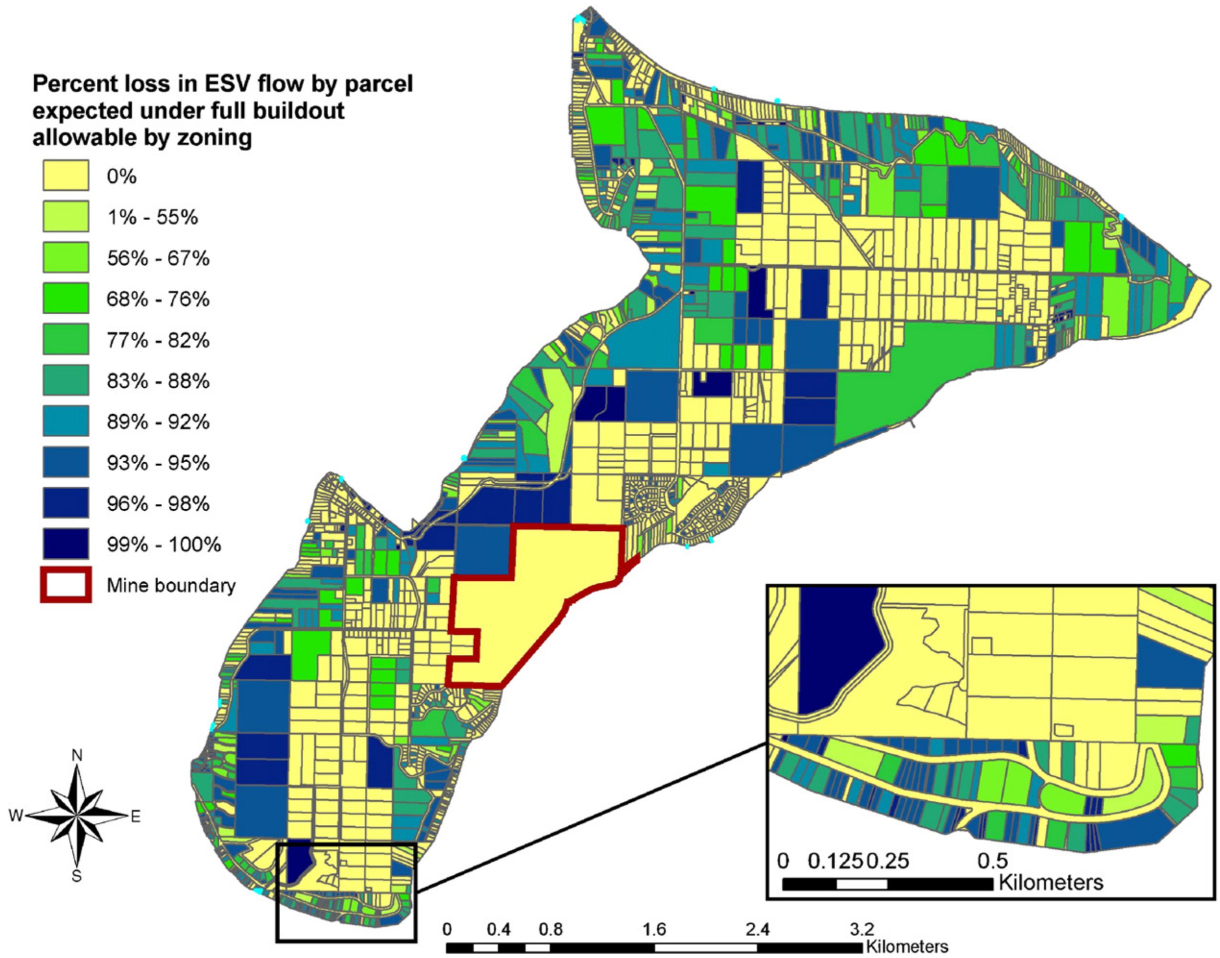

Fig. 5 – Estimated percentage reduction in yearly ecosystem service value flows between current conditions and full zoning buildout conditions by parcel for Maury Island in 2004 dollars.

Similarly, in the case of recreational and aesthetic ecosystem service values, the marginal social cost of losing 1 ha of Central Park in New York is likely to be far greater than that of losing 1 ha of otherwise similar green space in a rural area of upstate New York where green space is abundant (Fausold and Lilieholm, 1999).

A further factor complicating implementation of this framework is the availability of spatial data. Even within the United States, which has among the best publicly available spatial data catalogues in the world, the availability and quality of this data are highly variable by region, because of the role of state and local governments in developing fine-scale land cover data. While there are nationally available land cover data sets, such as the USGS's National Land Cover Dataset, their low spatial resolution, lack of categorical precision, and low classification accuracy for many cover classes limit usefulness for both local scale applications and projects where poorly classified types (e.g. agriculture) are of importance. While some recently released higher-resolution ancillary federal data sources, such as the National Hydrography Dataset and the National Wetlands Inventory can augment the NLCD and other national land cover products, in general, one can only rely on sufficient resolution and quality to conduct coarse scale applications requiring relatively low accuracy from nationwide land cover data.

The lack of spatial data availability is compounded by definitional challenges associated with land cover categorization. While some land cover categories like "wetlands" have a statutory definition, others are often very broadly defined and can include lands with highly diverse functional characteristics. For instance, a unit of land defined as "grassland" can be many things–pasture, hayfield, natural shortgrass or tallgrass prairie, savannah or golf course–all of which have very different functional profiles for delivering ecosystem services. Hence, when a valuation study is associated with a particular land use or land cover type, it is crucial to define that association as precisely as possible. Often, however, the analyst is faced with the decision of combining different classes together (e.g. applying a study of shortgrass prairie to a generic "grasslands" category) or of having no valuation estimates at all. In this case it is up to the analyst to judge which is the lesser of two evils within the context of the project and how this decision must be reported. In some cases these functional differences will be due less to biophysical differences than to socioeconomic ones. For instance, if a study estimating the recreational value of conifer forests was done on public land, it may be inapplicable for transfer to otherwise similar conifer forests on private land, where access, expectations and long-term management may be different.

These problems have led to gaps in landscape valuation. Such gaps can mislead users if the limitations are not made explicit. Where a particular ecosystem service associated with a particular land cover is not valued in the literature and no reasonable proxy exists, a value of zero must be assigned, just as we did for desert ecosystems in San Bernardino County. In such cases, the lack of transferable studies clearly results in a significant undervaluation, since there is no question that systems like desert scrub and arid woodlands provide very important services. Therefore, when significant gaps exist in a study, results should be treated as conservative lower bound estimates.

These challenges were at least partially surmountable in the aforementioned case studies. However, in many potential applications the ESV transfer method may be far more difficult or infeasible. As an example, large spatial extents present significant valuation problems. One reason for this is because valid estimation of the value of ecosystem service (which are supposed to reflect willingness to accept compensation for loss of those services) requires marginal analysis (Daily, 1997; Pearce, 1998)—that is, evaluating the effects of very small changes in the quantity of natural capital on welfare measures. One of the problems with evaluating ecosystem service values across large areas (like a continent) is that if the aggregate value represents the willingness for consumers to be compensated for the loss of all the natural capital in the study area, this would imply a significant shift in the supply and hence the shadow price of all types of natural capital. This critique was directed at Costanza et al. (1997) in their attempt to value natural capital for the entire world, since price shifts are increasingly unpredictable as quantity shifts approach the global level (Pearce, 1998). Eventually, the opportunity cost of the entire world's natural capital becomes incalculable since the existence of all life depends on it. While the scale of potential changes to natural capital addressed in our case studies do not meet the strict definition of "marginal", economists have found that for relatively contained or localized extents, price functions for environmental amenities can be assumed to be constant (Palmquist, 1992). Another problem with increasing spatial extent is the increasing heterogeneity within the cover classes being valued. As the area being valued increases in size, encompassing increasingly varied ecological and human contexts, the assumption of transferability of value estimates within a cover class is weakened.

Finally, perhaps the most important lessons learned from these case studies relate to how to make use of the results. For results to have validity in a management context there must be transparency and meticulous documentation at every step. Otherwise the framework runs the risk of becoming just another decision making 'black box'. Further, clients must understand that ESV estimates alone should not be the sole basis for management decisions. Over-reliance on ESVs is tempting, because a single dollar metric is easier to communicate than multi-faceted qualitative results. Instead, ESV estimates should be used as merely one type of evidence amongst many (e.g. habitat assessments, ecological field studies, biological inventories, socio-economic research, etc.) in supporting management decisions. An example of this approach is the Maury Island project, in which we worked with teams of field biologists and aquatic scientists (Herrera Environmental Inc.) and socio-economic researchers (NEI, Inc) to provide the client with a wide range of data and analysis from multiple disciplinary perspectives.

8. Future directions

We anticipate that the spatial valuation framework described in this paper will be refined and improved as the empirical literature on economic valuation of ecosystem services grows and the availability of spatial data increases. The number of studies measuring the economic value of ecosystem services has increased dramatically over the last decade (Heal et al., 2005; Rosenberger and Stanley, 2006-this issue), resulting in greater levels of specificity and reliability in our efforts to quantify the value of key ecosystem services. With the release of global reports such as the Millennium Ecosystem Assessment (2003), we anticipate that this trend will continue.

Digital spatial data has also dramatically increased in quality and availability recently, particularly at the state level. Many states in the USA have mapped local land use and land cover at very fine mapping scales, making it much more usable for ESV exercises. The trend towards more categorically precise and fine-scale land cover/land use data development is likely to continue for a number of reasons. First, availability of high resolution multi-spectral imagery has increased. Second, new technologies, such as object oriented imagery classification, have enabled the automation of the classification of high resolution imagery which, until recently, had to be done through expensive manual digitizing. The increasing coverage of LIDAR (airborne laser altimetry), is also likely to have implications for the mapping of ecosystem services. LIDAR results in extremely high resolution terrain surfaces and can be used to derive measures predictive of ecosystem services, such as above ground biomass of trees (Drake et al., 2003), canopy heights, stand volume and basal area (Dubayah and Drake, 2000), above-ground carbon (Patenaude et al., 2004) and stream and coastal geomorphology (French, 2003; Lohani and Mason, 2001). The increasing quality and availability of fine-scale social, economic, regulatory, and infrastructural spatial data is also promising for future valuation efforts. Currently there are limited opportunities to make use of these data sets in value transfer because of the lack of sufficient contextual variation in the valuation studies, but as more valuation studies are conducted across a range of socio-economic, demographic and regulatory conditions, such data will prove to be highly useful.

By mapping ecosystems at higher levels of spatial and categorical precision and accuracy and linking them to reliable ecosystem service flow estimates, we can assist decision makers in the private sector and government as they seek to identify critical areas in the delivery of ecosystem services. Since any given location in the landscape can yield a bundle of ecosystem services, the challenge will be determining how to manage landscapes in a manner that maximizes the delivery of value to society while minimizing forgone market opportunities.

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