Assessing benefit transfer for the valuation of ecosystem services

Mark L Plummer

The valuation of ecosystem services can play an important role in conservation planning and ecosystem-based management. Unfortunately, gathering primary, site-specific data is costly. As a result, a popular alternate method is to conduct a "benefit transfer" (applying economic value estimates from one location to a similar site in another location). Among the potential pitfalls of such an approach, the correspondence (or lack thereof) between the locations is probably the most important for evaluating the probable validity of the benefit transfer. A common type of benefit transfer in ecosystem service valuation applies an estimate of value per hectare to all areas having the same land-cover or habitat type, and is particularly susceptible to errors resulting from lack of correspondence. Enhancing the use of benefit transfers in this and other ecosystem service applications requires paying closer attention to simple guidelines, developed by economists, for improving validity and accuracy.

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Near the town of Gramercy, Louisiana, on the edge of the Maurepas Swamp, there once was a pearl of great price: a bottomland hardwood wetland covering about 6 acres (~2.4 hectares). Not particularly distinctive and somewhat isolated due to encroaching development, nevertheless, those 6 acres were worth over $215 000, or about $35 000 per acre, according to a 1995 study (Breaux *et al.* 1995).

Attaching dollar figures to nature is increasingly popular, as cajoling people to save a swampy piece of bottomland forest – not for filling it in and putting up a shopping mall, but for keeping it in an undeveloped state and letting it function as Nature intended – is easier when large sums of money are mentioned. Indeed, projecting that value onto all 51.4 million acres of freshwater forested wetland in the US (Dahl 2000) produces a figure that would command a lot of attention: about $1.7 trillion.

In fact, such a projection produces nonsense. The source of the Gramercy wetland's value was not so much its intrinsic value as what was next door. Zapp's Potato Chips, maker of the "Cajun Crawtator", used the wetlands to treat its wastewater, a method officially permitted and endorsed by the Louisiana Department of Environmental Quality, and even potentially beneficial to the wetland (Day *et al.* 2004). The arrangement benefited Zapp's by lowering its wastewater treatment costs. The $215 000 figure is an estimate of how much Zapp's saved by treating its wastewater with a wetland. How much society at large benefited from that treatment is unknown, but the wetland produced a substantial (if private) benefit.

Not all freshwater forested wetlands have an adjoining potato chip factory, and so projecting Zapp's cost savings onto other, similar wetlands would be inappropriate. But there might be other sources of value, both private and social, that the Gramercy wetland and other wetlands share. For example, the Maurepas Swamp is home to white-tailed deer, American alligators, largemouth bass, and even bald eagles. These species provide wildlife-viewing opportunities and some provide outlets for recreational hunting (Figure 1), both of which have an important economic value. Under what circumstances would it be valid to project estimates of those values taken from the Maurepas Swamp onto another wetland?

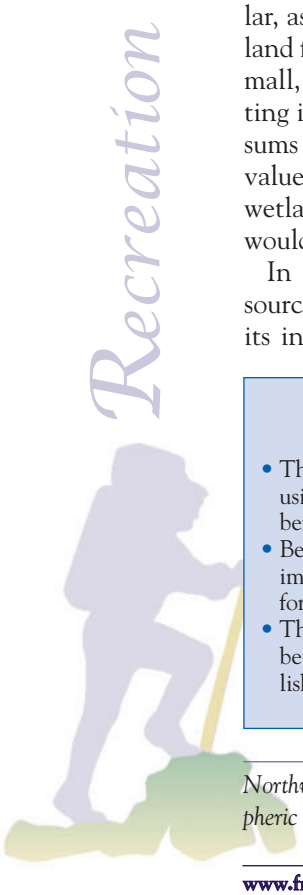
The answer to this question comes from considering the practice of benefit transfer, a technique economists use to take estimates of economic value from one site and "transfer" them to another site. This approach is finding increasing favor for the analysis of ecosystem services, one of which – "wastewater assimilation" – underlies the high value of the Gramercy wetland. Yet as that example illustrates, benefit transfer should not be pursued blindly. Not all ecological systems are pearls of great price.

In this paper, I examine how benefit transfer has been used for ecosystem service valuation, focusing on a com-

In a nutshell:

- The valuation of ecosystem services at one site is often done by using existing estimates from other sites, a practice known as benefit transfer
- Benefit transfer is subject to several possible errors, the most important of which occurs when existing sites are poor matches for the ecosystem under consideration
- The use of benefit transfer for ecosystem service valuation can be improved by paying closer attention to guidelines established by economists for the use of this method

Northwest Fisheries Science Center, National Oceanic and Atmospheric Administration, Seattle, WA (mark.plummer@noaa.gov)



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mon method that applies an estimate of value per hectare to all areas having the same land-cover or habitat type. As discussed below, this approach is vulnerable to error stemming from a lack of correspondence among the sites considered. Using guidelines that economists have developed for more traditional uses of benefit transfer, I describe how its application to ecosystem services can be improved.

The practice of benefit transfer

Benefit transfer is a procedure for taking the estimates of economic benefits (or values in general) gathered from one site and applying them to another. The site from which the estimates are taken is called the *study* site, in that it is a site that has already been studied in some way. The site to which the estimates are applied is called the *policy* site, because benefit transfer is usually part of an economic analysis of a proposed policy action. Benefit transfer is rarely the best choice for analyzing the economic value of a policy, but the costs of gathering primary, site-specific data have made it a common practice for studies of the recreational uses of natural sites (Rosenberger and Loomis 2001; NRC 2005).

For environmental applications such as assessing ecosystem service values, the focus is on a policy that changes the extent, or other biological and physical characteristics, of a natural site or area. "Value" refers to how much *people* value the change and is usually expressed as a monetary amount that they are willing to pay (EPA 2000; NRC 2005). For a particular individual, this willingness-to-pay depends on the biophysical characteristics of the site and other relevant economic and social measures, as well as the extent and nature of the changes brought about by the policy. Finding the aggregate economic value of a policy is then a matter of identifying the individuals affected by the change and aggregating the individual willingness-to-pay amounts over that population (Smith 1992; NRC 2005). From an economic perspective, then, value is not intrinsic to a particular site or ecological system. It must be evaluated in the context of specific biophysical and human characteristics (Bockstael *et al.* 2000).

Ideally, measuring these values would rely on data gathered directly from the policy site and the population affected by the policy. Because of the time and expense of gathering such primary data, however, benefit transfer is often viewed as an acceptable substitute. A proper benefit transfer usually consists of three steps, following guidelines economists have developed to improve this practice (EPA 1993, 2000; Rosenberger and Loomis 2001). First, the analyst carefully describes the policy site and the proposed policy action(s). The description should specify the important biological and physical characteristics of the site, and how humans are expected to use the site (eg fishing for recreational species) or have a connection to it in "non-use" ways (eg valuing the existence of an endangered fish population that inhabits the site). It should also identify the extent of the human population affected by the policy.

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Figure 1. *Maurepas Swamp. This area, a mixture of prairie and swampland, is popular for hunting waterfowl.*

The description produced in the first step effectively creates criteria to guide the second step, in which the analyst selects suitable existing studies to provide a basis for a benefit transfer. Of obvious importance is whether an existing study covers the same type of uses or non-use connections that are affected by the policy under consideration, and similar types and extents of changes that are the basis for the study valuation. In general, the literature is much more extensive for economic values derived from uses such as recreation than for values derived from non-use connections to a site, making benefit transfer much easier for the former. Equally important is whether the study site's characteristics are similar to those of the policy site. The degree to which all of these characteristics of existing studies are similar to those of the policy site determines what is called *correspondence*, which is central to determining the accuracy of a benefit transfer (Rosenberger and Phipps 2007). The studies chosen should also, of course, meet the usual data quality conditions: adequate data, sound economic method, and correct empirical technique.

From the set of studies that are judged to be sufficiently similar, the analyst then derives an estimate of the economic value of the relevant use or connection, and applies it to the policy site. A common method for this last step is to use a unit value: a dollar estimate of economic value on a per-unit basis, taken from one or more studies, where the unit can be based on an activity (eg per fishing day) or an outcome (eg per fish caught), or on a per-person basis. In any case, the unit value is usually expressed as a constant per-unit amount or as a range of constant per-unit amounts. The constant value (a single amount or each endpoint of a range) is then multiplied by the projected quantity of use at the policy site, or by the projected number of people who hold non-use values connected to the site.

An alternate (and preferred) approach is to use a benefit function (Loomis 1992). A benefit function relates an individual's willingness-to-pay to a set of individual and site characteristics. Such a function can be based on one

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study site, in which case important biophysical characteristics will not vary and so cannot be included in the function (Ready and Navrud 2005). If a diverse set of studies is available, the benefit function can be estimated with a meta-analysis of the study results, in which case characteristics that are constant with respect to any one study, but vary across the studies, can be incorporated into the function (Smith and Pattanayak 2002; Shrestha and Loomis 2003; Bergstrom and Taylor 2006; Hoehn 2006). The benefit transfer then takes place by measuring the function's variables at the policy site and evaluating the function at those values.

Ideally, the available studies for benefit transfer would be rich enough to include studies that cover sites, policies, and human populations identical to those of the policy site (Boyle and Bergstrom 1992). The studies themselves might suffer from measurement and other errors, but the act of transferring the estimated values would not substantially increase the estimate's error. The real world rarely offers such ideal conditions, however, and so the art of benefit transfer lies in finding ways to minimize transfer errors, while not expecting to eliminate them altogether.

The most important source of transfer error lies in the problem of finding studies of similar sites, or in meeting correspondence. The failure to adequately meet correspondence is called *generalization* error and stems from viewing study and policy sites as members of a more general group of sites, based on a few characteristics, and (mistakenly) treating them as equivalent in *all* of their characteristics. For gauging the correspondence of two (or more) sites, economists have traditionally focused on characteristics of the human population, such as income and other socioeconomic measures, but biophysical site characteristics are also important (Rosenberger and Stanley 2006; Rosenberger and Phipps 2007). For estimating the value of a hiking trail, for example, a study site could have spectacular views, whereas a policy site might have only average views, or vice versa. Transferring the constant value per day of hiking from the study site to the policy site would likely over- or underestimate the policy site's value, respectively. If a benefit function can be constructed, differences in the site characteristics and human populations between the two sites can be accounted for by using the function, which will reduce generalization errors (Loomis 1992; Kirchhoff *et al.* 1997; Van den Berg *et al.* 2001).

Benefit transfer is an expedient way of producing estimates of economic values when primary, site-specific data are lacking, but it will always be a "second best" valuation method (NRC 2005). Nevertheless, the practical need to assess economic values in wide-ranging natural locations has produced numerous studies that use the technique.

Benefit transfer and ecosystem services

Using the benefit transfer procedure for ecosystem service valuation is conceptually straightforward. Ecosystem service valuation consists of four steps (Freeman 2002): (1) determining how, and how much, a policy will change ecosystem structure and function; (2) determining how these changes will affect the flow of ecosystem services; (3) placing an individual value (willingness-to-pay) on these service changes; and (4) aggregating the individual willingness-to-pay values over the population affected by the ecosystem service changes. It is in the third step that benefit transfer can be used.

How have ecosystem service valuations actually used benefit transfer? A comprehensive survey is beyond the scope of this paper, but an increasingly common use is one that measures the quantity of ecosystem services with spatial data, and then uses benefit transfer to quantify their monetary values. Known as ecosystem service mapping, this approach estimates ecosystem service values over a landscape in the following way (Troy and Wilson 2006). First, a particular area (eg a state, county, or ecological region) is differentiated by land cover, biome, or some other set of ecologically based landscape types. Drawing from a standard set of ecosystem service categories (eg de Groot *et al.* 2002), each landscape type (eg forestland) is then linked to a set of services (eg recreation or carbon sequestration) believed to be provided by that type. At this point, original valuation data can be gathered for each type of service and landscape. More commonly, a benefit transfer exercise is conducted, drawing on studies that meet two criteria: (1) the study estimates an economic value for one or more types of activity or ecological function that can be linked to a particular ecosystem service category, and (2) the study site can be linked to a particular landscape type.

The next step is to take the study's estimated value for a particular ecosystem service and divide this by the area of the relevant landscape type, producing a constant value for that ecosystem service–landscape type combination per unit of area (eg recreation or carbon sequestration value per acre of forestland). The total value of this service for a landscape type is then found by multiplying the unit value by the acreage of that type, a calculation similar to the one done for a traditional benefit–cost exercise based on unit values (Figure 2). The aggregate ecosystem service value for the entire area is then found by summing over all services and landscape types.

The use of benefit transfer for this type of ecosystem service valuation began with what is, to date, the largest benefit transfer exercise ever conducted: the economic valuation of the entire planet (Costanza *et al.* 1997, but see Toman 1998, Bockstael *et al.* 2000, and Freeman 2002 for criticisms). Covering 17 ecosystem services for 16 biomes, the paper cobbled together estimates from over 80 economic studies, supplemented by some original data. The use of benefit transfer was almost universal, as even the original data used in the study were gathered in one or more locations and then projected worldwide.

Using benefit transfer in exercises such as Costanza *et al.* (1997) raises serious issues about the accuracy of the transferred values. Because ecosystem service mapping

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uses a single characteristic to match study and policy sites, correspondence rests on whether other relevant characteristics of the study site(s) are similar to the same characteristics for the policy site – that is, for every pixel or polygon of a particular land-cover/use type over the landscape under consideration. For some characteristics, adequate correspondence might be possible if the landscape scale is small enough – a watershed, for example. Even on this scale, however, generalization error can still be significant, as it would be if the value attributed to the Gramercy wetland was transferred to wetland pixels or polygons that lacked an adjacent potato chip factory. Therefore, without a diligent filtering of study sites, or a substantially less ambitious policy landscape, the potential for generalization error in an ecosystem service mapping exercise may be substantial.

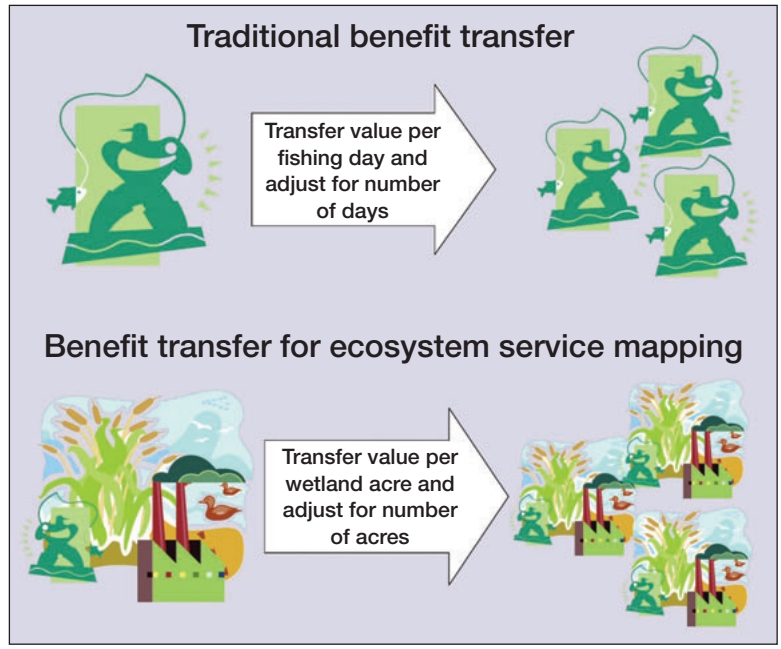

Figure 2. *A traditional benefit transfer using a unit value derives the value from one or more studies of a particular type of site and activity (eg fishing at a freshwater lake). This value is then multiplied by the expected level of activity at the policy site. For ecosystem service mapping, a benefit transfer derives a unit value from studies of a particular ecosystem type and service (eg recreation or wastewater assimilation for a freshwater wetland). The transfer exercise multiplies this value by the area of the ecosystem type in the landscape under consideration.*

Two examples underscore this conclusion. The first is from Costanza *et al.* (1997), in which several studies are used to estimate an average value for the ecosystem service of "recreation" in the land-cover type of "tropical forest". Among these are two studies of the economic value of visitations ("recreation") to the Monteverde Cloud Forest Reserve, a popular ecotourism destination in Costa Rica (Tobias and Mendelsohn 1991; Echeverria *et al.* 1995). But the Reserve has human-built features that are not typical of the average tropical forest: dozens of hotels, several independent restaurants, a bank and supermarket, and even a cheese factory (Figure 3). Similarly, another "recreation" study used in Costanza *et al.* (1997) covered ecotourism in the Galapagos Islands, where, in the late 1980s, "luxury liners" and other vessels brought over 30 000 visitors annually to view the islands' wildlife (Edwards 1991).

A second example comes from Batker *et al.* (2008), a study of ecosystem service values in Puget Sound that used benefit transfer in a mapping exercise. One of the land-cover/use types included in this study is beaches, which is often the most valuable type on a per-acre basis (King County 2004; Costanza *et al.* 2007; Batker *et al.* 2008). Their study estimated values for three ecosystem services associated with beaches: disturbance prevention, aesthetic and recreational services, and cultural and spiritual services. The corresponding benefit transfer included studies of beaches in South Carolina (Figure 4a) and New Jersey (Figure 4b).

What is the likely correspondence between these beaches and beaches in Puget Sound, such as Alki Beach (Figure 4c)? Merely categorizing an area by its land-cover/use type – beach – ignores important natural determinants of the provision and value of the ecosystem services flowing from that area. In terms of disturbance prevention, the beaches in South Carolina have experienced seven hurricanes, two tropical storms, and several other severe weather events since 1993, while in that same period, Alki Beach and other beaches in Puget Sound have experienced hardly any of the severe weather events that underpin the value of this service (NOAA, no date). Without disturbances, what is the value of prevention? Similarly, over the summer months, the New Jersey beach has average water temperatures that range between 7° and 16°F (~ 4° and 9°C) warmer and average air temperatures that are about 8°F (~ 4.5°C) warmer than those of Alki Beach (NOAA 2002). The recreational services provided by a beach are generally enhanced by warmer water and air temperatures (Morgan *et al.* 2000). For both services, therefore, serious questions can be raised about the correspondence between the study site beaches and beaches in Puget Sound.

These examples are not conclusive, of course, but illustrate well the types of errors that can beset benefit transfer when correspondence is based on a single characteristic, such as land cover/use. Such potential errors might have the effect of convincing people that benefit transfer will *never* be successful when used in this way. That conclusion would be correct if the standard of practice is to commit no error, in which case benefit transfer would never be used in any application. But all empirical research produces results that contain some error, and so the proper response to these problems is to think of ways to make benefit transfer *better*, rather than perfect.

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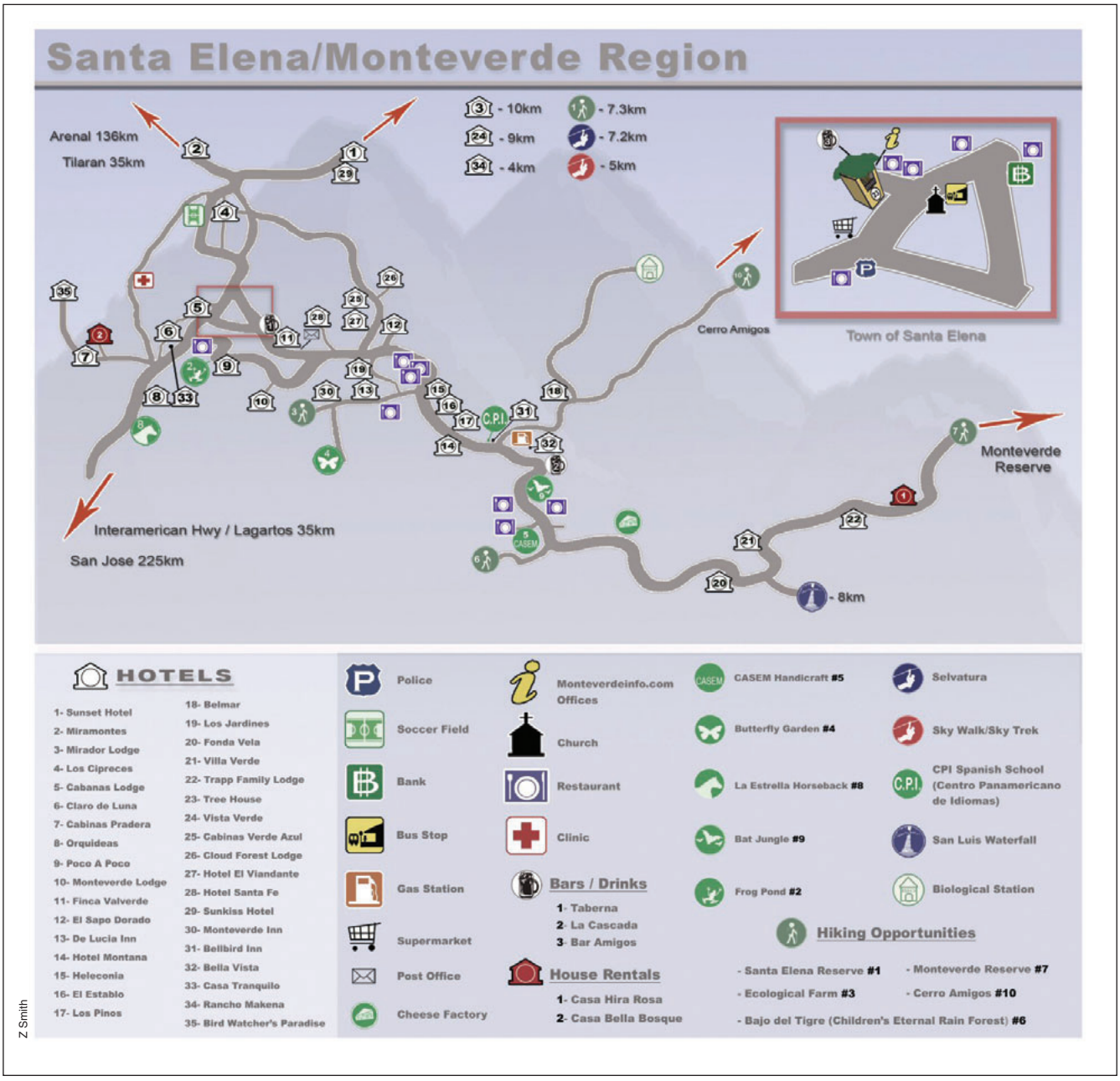

Figure 3. *The Monteverde Cloud Forest Reserve in Costa Rica provides ecotourism opportunities in a highly developed and accessible environment. Applying economic values of these opportunities to most tropical forest areas is likely to produce generalization errors.*

Improving the practice of benefit transfer for ecosystem service valuation

The desire to use benefit transfer for the measurement of ecosystem service values is motivated in part by a concern that unmeasured values are effectively treated as having zero value (Troy and Wilson 2006). In some cases, of course, ecosystem service values can be easily measured with local data, satisfying the concern and rendering a benefit transfer unnecessary. Many provisioning goods and services, for example, are sold in markets, and economists have straightforward ways of estimating value in these cases (Freeman 1993). Similarly, a global ecosystem service such as carbon sequestration is provided on a scale that effectively equalizes the value across all sites that provide this service, again rendering a benefit transfer unnecessary.

Not all ecosystem services fit into these categories, and so benefit transfer is still a desirable way of assessing their value. Nevertheless, arguing that "some number is better than no number" does not shield its use from criticism, just as economists have leveled criticism at its more traditional uses. Analyzing the accuracy of traditional benefit transfer exercises is an ongoing enterprise (see Rosenberger and Phipps 2007 for a recent, comprehensive review), as are efforts to improve its practice.

A first step in improving the use of benefit transfer for ecosystem service valuation, then, is to ask the question:

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Figure 4. *Two beaches have been studied for their economic values and used as the basis for benefit transfers in ecosystem service mapping exercises: (a) Surfside Beach, SC (Silberman* et al. *1992), and (b) Monmouth Beach, NJ (Pompe and Rinehart 1995). Transferring values from these beaches to beach sites in Puget Sound, such as (c) Alki Beach, WA, risks generalization error if the study site and policy beaches have poor correspondence for characteristics other than their common land-cover type.*

how accurate are ecosystem service values generated by benefit transfer exercises? Addressing this question would be particularly valuable for ecosystem service mapping, which a priori seems highly susceptible to generalization error. Documenting the magnitude of these potential errors would better enable policy makers to evaluate the tradeoffs between the shortcomings of benefit transfer and its substantially lower cost. For ecosystem service mapping, such a test would require a comparison of value estimates based on primary data with those generated via spatial data and benefit transfer. Although such a test may be too costly on the typical scale of such mapping exercises, it might be possible for an area such as a watershed, for a limited number of ecosystem services.

The practice of benefit transfer for ecosystem service valuation can be improved in other ways, although pursuing these may serve to reduce its usage. Explicitly addressing the basic elements of a benefit transfer exercise would make the efforts more transparent and, with particular attention to the problem of correspondence, more defensible. This approach could include the following elements.

Better characterization of potential policy actions that affect ecosystem services

Valuing a limited set of ecosystem services at a single site with well-defined, natural boundaries can accomplish this easily. A mapping exercise that attempts a comprehensive valuation of many services over a broad landscape faces a greater challenge in doing so. Nevertheless, for ecosystem service valuations to be useful to policy analyses, we must ensure that the policy embedded in the benefit transfer exercise is the right one.

Better characterization of how the policy being considered affects ecological structure and function, and how those effects will change the flow of ecosystem services

In some cases, the policy framework will be all-or-nothing – a wetland will be filled (entirely) and replaced with a parking lot, for example. Most policies fall short of this extreme, and so their effects on ecological conditions need to be carefully described. The second part is equally important and requires addressing the issue of metrics:

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"simply defining what to measure and how to measure it" (Kline 2007). These metrics can become the foundation for building benefit functions and unit values for ecosystem service benefit transfer.

Better identification of the beneficiaries of ecosystem services

The population affected by a policy change that alters the flow of ecosystem services will depend on the type of service. Services that are actually marketed; services that are non-market in nature but consumed on-site; services that are non-market in nature and produce ecological effects off-site – because each will have different affected populations, care is required in aggregating individual ecosystem service values.

None of these suggestions will eliminate generalization and other types of benefit transfer errors. Ultimately, the best way to reduce transfer errors is to build a better collection of studies, which use primary data to value ecosystem services in ways that lend themselves to benefit transfer. For the time being, however, paying closer attention to the traditional guidelines for benefit transfer may lessen its use for ecosystem service valuation, but can increase its validity and accuracy.

■ Conclusions

In the early 1990s, sales of Zapp's potato chips took off, and with an expansion of their business came an increase in the volume of their wastewater. The costs associated with using the Gramercy wetlands grew too high, and so around 1993 the company discontinued that form of wastewater treatment. The pearl of great price was no more (D Powell, interview with M Bruin, Zapp's Potato Chips).

Bringing the story of Zapp's Potato Chips and the Gramercy wetlands up to date does not undercut the importance of ecosystem service valuations. Those values are real and potentially important, and they should play an important role in the analyses that help inform conservation policies. Instead, the story underscores the importance of addressing the correspondence issue for benefit transfer exercises involving ecosystem services, because those values can be significant but highly variable, causing substantial generalization error. Indeed, the value of the Gramercy wetland may change again. Zapp's is working with the town of Gramercy to revive the wetland treatment option, so the wetland's value may eventually be higher than ever (D Powell pers comm).

The use of benefit transfer for ecosystem service valuation exercises should be an area where economists and others tread carefully, but they should not allow their caution to inhibit such efforts altogether. The "first best" solution will always be the collection and use of primary, site-specific data. Absent the resources to do this, benefit transfers can provide insights into the values of ecosystem services. Its use in this area, however, requires closer attention to the sources of potential error and the adoption of better practices to address issues such as correspondence. Doing so will increase the validity and accuracy of ecosystem service values estimated in this way.

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Research Emphasis: We invite applications from prospective Ph.D. students with research interests in land use and land cover change, responses of plants, arthropods and vertebrates to ecological disturbances, and variability of parasites in relation to changes in vertebrate population density. Our research focus uses tools such as remote sensing, geographic information systems, telemetry, systems modeling, molecular genetics, and other analytical resources to assess how northern bobwhites and grassland vertebrate populations vary in response to land management actions and habitat manipulations.

Research Support: The Institute provides state-of-the-art research facilities and equipment as well as competitive financial support in the form of research assistantships or endowed fellowships. We provide excellent opportunities for graduate student education in an exciting and dynamic program that focuses on both field and laboratory research.

Start Date: March or September 2009

Applicants should have a background in ecological or natural resources, environmental engineering, or wildlife sciences.

For more information, please contact:
Dr. Leonard A. Brennan, *Endowed Chair for Quail Research*
at leonard.brennan@tamuk.edu

www.ckwri.tamuk.edu

TEXAS A&M UNIVERSITY - KINGSVILLE

MAKE THE MOST OF YOUR VALUABLE DATA!

Online Course – Maximum Likelihood Estimation for Natural Resources and Ecology

This online non-credit course offered by the Quantitative Fisheries Center at Michigan State University will introduce fisheries biologists and ecologists to maximum likelihood estimation. The course covers nonlinear model fitting, estimation of uncertainty in model estimates (e.g., likelihood profiling), and methods of inference using multiple models (e.g., AIC model averaging). The course complements materials presented in the "The Ecological Detective" by R. Hilborn and M. Mangel, and is self-paced, with worked Microsoft® Excel exercises using ecological and fishery related data.

QFC Enrollment cost: $470 per person. Please go to http://qfc.fw.msu.edu/courses_mle.asp to learn more or to enroll.