The Journal of
Regional
Analysis
& Policy

JRAP 42(3): 188-197.

Economic Value of Stream Degradation across the Central Appalachians

Laura E. Jackson[#], Brenda Rashleigh[*], and Michael E. McDonald[#]
U.S. Environmental Protection Agency Office of Research and Development, National Health and Environmental Effects Research Laboratory ([#]Research Triangle Park, NC; [*]Narragansett, RI) – USA

Abstract. This study demonstrates a method to calculate the economic value of the loss of a highly valued ecosystem service—the provision of recreational fishing—across a multi-state assessment region. We estimated annual freshwater fishing expenditures foregone from degraded conditions in wadeable streams that are potential habitat to one or more of four sportfish species. Using probability-based federal surveys for data on sportfish presence, we developed range models for the four species in the mountainous portions of four U.S. mid-Atlantic states based on geophysical stream variables unrelated to habitat condition. From these models, we determined the proportion of the wadeable stream resource (44.2%) that could potentially host sportfish and allocated an estimate of annual regional freshwater fishing expenditures (US$826 million) from the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation to these stream segments. We attributed the absence of sportfish in these segments to stream degradation; an additional US$239 million was estimated as lost freshwater fishing expenditures. These figures suggest a considerable annual economic incentive for stakeholders to restore and protect stream habitat for the maintenance of sport fisheries. This method is readily transferable to other U.S. regions where long-term surveys that collect metrics linked to ecosystem services are in place.

1. Introduction

Most regions of the country have a history that was forged by past economic and environmental decisions. The economic viability of a region depends on the arrival or departure of industries which enhance opportunities for economic stability. Highly desirable regional natural resource attributes can form the basis for the development of a strong tourism industry. This can lead to a more stable regional economy, both directly through tourism and through amenity-based development that depends on the natural resources to attract and retain people and businesses. However, if the quality of these natural resources declines and is not restored, this is the equivalent of the departure of a substantial industry from the region.

A considerable body of research has developed to communicate environmental stewardship issues in economic terms (e.g., Costanza et al., 1998; Wilson and Carpenter, 1999; Ricketts et al., 2004; National Research Council, 2005). Assigning monetary value to natural resources such as wildlife species (e.g., Cleveland et al., 2006; Losey and Vaughn, 2006) and native habitats (e.g., Pearce, 2001; Nunez et al., 2006; Costanza et al., 2008) raises the profile of the natural world in decision-making and facilitates comparisons with competing concerns. The concomitant focus on ecosystem services emphasizes the ways in which functional ecosystems are essential to human health and prosperity (e.g., Daily, 1997; Foley at al., 2005). Many have argued (e.g., Ludwig, 2000; Hall and Klitgaard, 2006; Kumar and Kumar, 2008) that monetary valuation of ecosystem services is inappropriate or at least incomplete, as this approach is anthropocentric, cannot address moral or spiritual imperatives, and fails to capture even utilitarian benefits that are external to economic markets. Yet Earth-centric arguments have proven insufficient to stay the forces of human population expansion and desire for material comfort and pleasure (Kareiva and Marvier, 2007). For ecosystem services with significant market potential, monetary valuation remains a persuasive means by which to justify environmental stewardship, and methods are needed for this valuation (Daily and Ellison, 2002; Salzman, 2006).

We examined nature-based recreation, a frequently-monetized provisioning service (e.g., Walsh et al., 1984; Shafer et al., 1993; Stoll et al., 2006), albeit with potential negative consequences of overuse. We focus here on recreational fishing because of its ubiquity, established valuation methods, plausible linkages to available data on aquatic populations and water quality, and economic benefits to local communities for maintenance and restoration. In the U.S., fishing is one of the most favored leisure activities, nature-based or otherwise, scoring 9.1 on a scale of zero to ten and trailing only playing sports (9.2) and sex (9.3) in a 1985 national survey (Robinson and Godbey, 1999). Recreational anglers contributed US$35.6 billion to the U.S. economy in 2001 (U.S. Dept. of the Interior and U.S. Dept. of Commerce, 2002). As a point of comparison, this figure is more than the amount (US$32.8 billion in 2000 dollars) that the U.S. motion picture and sound recording industry added to the gross domestic product in the same year (U.S. Dept. of Commerce, 2007).

Numerous approaches are used to value nature-based recreation, including assessments of economic impact and value and estimates of social welfare, or consumer surplus. Revealed and stated preference methods are commonly used to quantify changes in social welfare associated with a change in ecological condition or management. Shafer et al. (1993) compared these methods using travel cost and contingent valuation to estimate the societal value of recreational activities, including fishing, in Pennsylvania. Ojeda et al. (2008) and Weber and Stewart (2008) used contingent valuation to value fisheries along with other services in riverine systems. Loomis et al. (2000) used contingent valuation to estimate the value of restoring multiple ecosystem services for a section of river.

Economic impact assessments evaluate the effect of recreational spending on local economies by tracing spending through purchases in order to add up the multiplier effects that such spending has on job creation or economic output. For example, Knowler et al. (2003) used a bioeconomic model to quantify the economic impact of a salmon fishery and compared differences in impact under different land-use alternatives. Another approach to resource valuation is to estimate recreational expenditures as a measure of direct benefits to the local community, or of value lost when the resource is removed or damaged. The Nature Conservancy (2009) valued Florida's environment at $8 billion annually, using only in-state fishing and hunting expenditures. Dodds et al. (2009) used cost estimation, including recreation and angling costs, to assess the annual economic damage of freshwater eutrophication across the US. The use of methods to estimate value lost from resource degradation, as opposed to current value, can inform decisions about resource restoration and management.

Here, we use recreational expenditure data to estimate the value of sport-fishing opportunities across the central Appalachian region of four eastern U.S. states and the value lost due to stream impairment. We focus on a geophysical region, because fishery resources, fishing opportunities, and human preferences vary by geophysical setting. Sportfish in the central Appalachians include smallmouth bass and three species of trout, two of which are non-native. Both native and introduced species are managed though stocking; geophysical variables drive their potential distribution. In the southern Appalachians, Flebbe et al. (2006) used elevation and latitude, and Schmitt et al. (1993) used gradient, elevation, and stream width, to identify potential habitat for all trout and brook trout, respectively. Argent et al. (2003) used drainage basin, physiographic province, median watershed slope, and watershed stream size to develop habitat profiles for all fish species in Pennsylvania. Drawing on these methods, we estimated distributions for sportfish species across the central Appalachians given healthy fisheries habitat and attributed any absences from seemingly suitable streams to degraded conditions. Next, we used the estimation of recreational expenditures approach to assess the value lost from these degraded streams and summed to create a total for the region. Consistent, high-quality survey data enabled region-wide inferences of the extent to which stream degradation has limited the ecosystem service of sportfish provision and the resultant loss to the region of recreational expenditures.

2. Study area, ecological data, and fish models

Our study area was the mountainous portions of Maryland, Pennsylvania, Virginia, and West Virginia (approximately 77,000 square miles, or 200,000 km^2, Figure 1). Biological and other data were collected from stream segments across this region through the U.S. Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP). This program initiated the ecological evaluation of stream condition with a probability-based field survey that includes metrics of fish and benthic macroinvertebrate populations, geophysical characteristics, and habitat structure in first- to third-order wadeable streams (Lazorchak et al., 1998; Davis and Scott, 2000; U.S. EPA, 2000). EMAP data and metadata are available at www.epa.gov/emap.

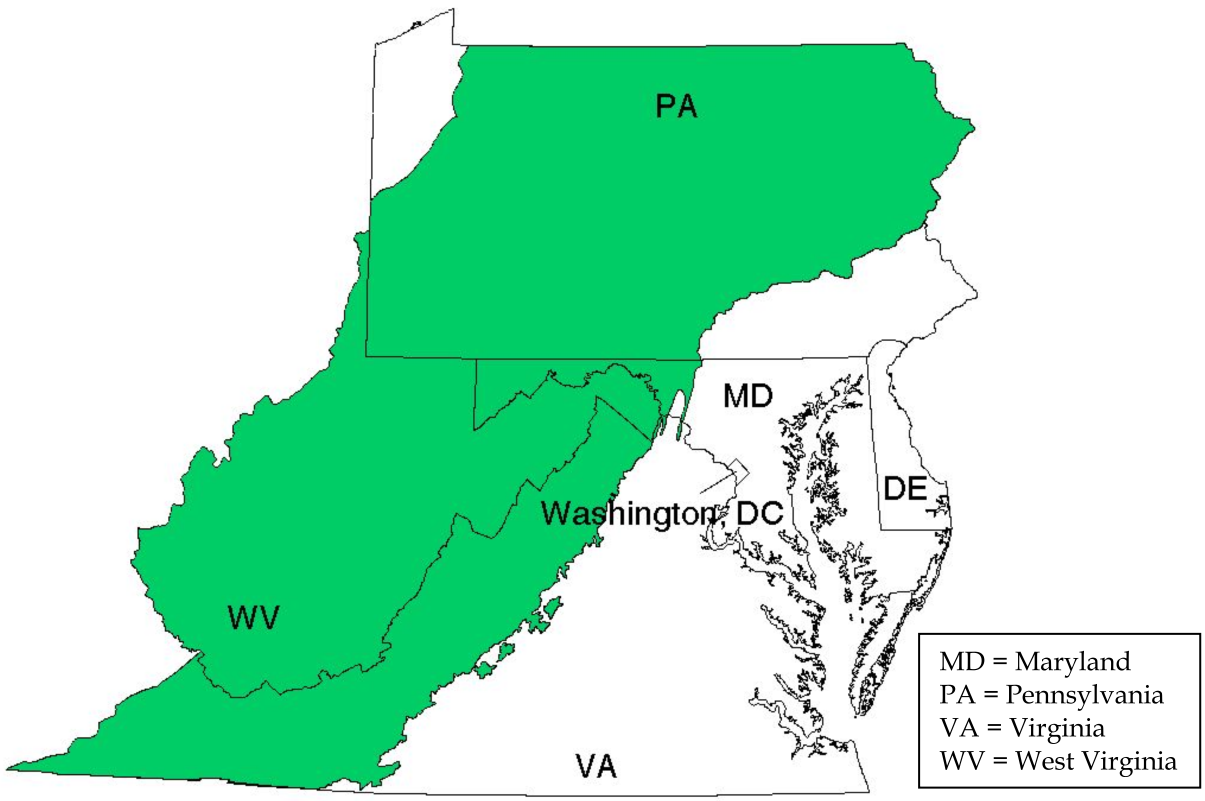

Figure 1. The central Appalachians study area.

We analyzed occurrence data for the four recreationally important species—brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*) and smallmouth bass (*Micropterus dolomieu*)—as well as segment geophysical and condition variables. For each sportfish species, we developed generalized boosted regression models (GBM) to predict the presence of a species at a site (Ridgeway, 2010). GBM is a machine-learning method that combines a boosting algorithm and a classification tree algorithm to construct an ensemble of trees. Default values were used and the optimal number of iterations was chosen using 5-fold cross validation. We selected elevation, latitude, slope, and flow (as a proxy for stream width) as predictor variables and allowed two-way variable interactions in the model (Table 1). These variables are distinct from stream condition and reflect those used in similar studies (Schmitt et al., 1993; Argent et al., 2003; Flebbe et al., 2006). The Receiver Operating Characteristic (ROC) curve describes the performance of a model across the entire range of classification thresholds. We use the area under the ROC curve (AUC) as a summary statistic of diagnostic performance for the models (Elith et al., 2008). The higher the AUC, the better: AUCs between 0.7 and 0.8 are considered acceptable; AUCs between 0.8 and 0.9 indicate excellent performance, AUC=1 would be perfect prediction.

Table 1. Number of sampled wadeable stream segments with and without sportfish, by state.

State	Number. of sampled stream segments	Number with sportfish	Number without sportfish
Maryland	7	3	4
Pennsylvania	67	42	25
Virginia	66	28	38
West Virginia	36	20	16
TOTALS	176	93	83

Data taken from U.S. EPA Environmental Monitoring and Assessment Program (U.S. EPA, 2000).

In order to transform the results of species distribution modeling from habitat suitability to presence/absence distribution, we used a prevalence of model-building data as the threshold (Liu et al., 2005). The model was tested on fish surveys from 176 statistically-representative stream segments in the central Appalachians sampled during 1993-1994. We made the assumption that stream segments meeting the model criteria for at least one sportfish species, but lacking the fish in samples, were likely in a degraded condition and represented a loss of economic value to the region.

We supported inferences of degraded stream condition with scores for a benthic index of biotic integrity (BIBI) (Davis and Scott, 2000). Benthic indices of biotic integrity are well-recognized indicators of ecological condition for streams (e.g., Maxted et al., 2000; Butcher et al., 2003). We tested for a statistically significant difference between the mean BIBI values for stream segments with sportfish present and for those without them but potentially suitable. A significant difference in BIBI values would support the assumption that sites lacking sportfish are more degraded than those where sportfish are present.

3. Recreational use and benefits data

Once we identified degraded streams within the region, we used the estimation of recreational expenditures approach to assess the value lost due to sportfishing from these degraded streams and summed to create a total for the region. We derived the economic value of the sportfish stream resource in the study area from the 1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Dept. of the Interior and U.S. Dept. of Commerce, 1998). Conducted approximately every five years since 1955, this telephone survey queries respondents on their in-state and out-of-state participation in wildlife-associated recreation and expenditures by type. These data are statistically representative at the state level and comparable across states. We extracted total in-state trip and equipment expenditures for all freshwater fishing, excluding the Great Lakes, by U.S. residents for the four states in the study area. For each state, we amended this figure to reflect the percentage of wadeable stream length that falls within the central Appalachian study region (Horn and Grayman, 1993; Herlihy et al., 2000). We attributed all reported freshwater fishing expenditures (excluding the Great Lakes) to wadeable mountain streams.

We attributed the derived 1996 regional expenditures in the study region to the proportion of wadeable streams represented by the EMAP segments where field crews recorded at least one occurrence of a sportfish species during the 1993-1994 survey. We assumed that streams with potentially suitable sportfish habitat but no sportfish recorded represented lost recreational revenue. Using our estimated regional proportion of degraded streams, we calculated the additional dollars that would have been spent if all of the potentially suitable streams had sportfish present. As a conservative measure, we further applied a ratio of 0.67 to 1 for recreational demand in response to increased sportfish habitat supply. This adjustment accounts for reported estimates of fishing behavior in response to increased resource availability (Loomis, 2006). We were able to calculate regional proportions of streams in satisfactory and degraded conditions for sportfish with known levels of statistical confidence, due to the EMAP probability-based sampling design. The proportion of total stream length represented by each stream segment varied due to unequal probability sampling weights (Stevens and Olsen, 2004).

4. Results

Of the 176 wadeable stream segments included in this study, 93 segments (representing 44.2% of the total stream length in the study region) had at least one occurrence of a sportfish species during sampling; 83 segments (representing 55.8% of the regional resource) had no sportfish present. The 95% confidence bounds for both regional estimates are +/- 8.5 percent. Table 1 lists the number of sampled segments by state. EMAP field crews recorded native brook trout in 40 segments from across the region but primarily in Pennsylvania; 28 of the 93 sportfish segments had two or more trout species present at the time of sampling. Small-mouth bass were present in 43 segments, only three of which were concurrent with observations of brook trout.

Reasonable GBM models could be developed for all four species of fish. AUC values for the species were as follows: *Micropterus dolomieu*, 0.89; *Oncorhynchus mykiss*, 0.80; Salmo *trutta*, 0.88; *Salvelinus fontinalis*, 0.83. Of the 83 stream segments with no sportfish present, 40 were predicted to be potentially suitable for at least one sportfish species. These represented 19.1% of the wadeable stream resource in the region—approximately 13,400 miles (21,527 km.) of stream. The mean BIBI value for the 93 segments with at least one sportfish occurrence was 63.1 (sd = 21.1), compared with a mean BIBI value of 44.0 (sd = 23.4) for the 40 segments with no sportfish present. A t-test indicated that these means are statistically different ($t = 4.47$; $p \sim 0$).

From the estimated 19.1% (+/-8%) of the study region failing to support sportfish in physically suitable habitat, we calculated US$239.2 million in foregone recreational freshwater fishing expenditures for the central Appalachians in the year 1996 alone (Table 2). Incorporating the margins of error from both surveys (42%; see below) results in an annual loss estimate of US$139 – 340 million. Converted to 2011 dollars with the U.S. Bureau of Labor Statistics inflation calculator (www.bls.gov), the estimated annual loss is US$199–487 million.

Table 2. Estimates of actual (columns 1-6), potential (column 7), and foregone freshwater fishing expenditures (column 8-9), and data used in calculations.

1. State	2. Total stream length (km)[a]	3. Stream length in study region (km)[a]	4. Contribution to study region	5. Total 1996 freshwater fishing expenditures, with margin of error (+/- S_e x 1.96)[b]	6. 1996 expenditures attributed to stream length in study region	7. Computed 1996 expenditures given sportfish in all potential habitat	8. Computed lost 1996 expenditures from degraded streams in study region	9. Reduction of net loss to 67% to account for diminished use with increased supply[c]
MD	17,798 *(9.8%)*	3,055 *(17.2%)*	2.7%	$112,856,000 *(+/-48%)*				
PA	65,196 *(36.0%)*	46,636 *(71.5%)*	41.4%	$ 468,441,000 *(+/-31%)*				
VA	65,464 *(36.2%)*	30,505 *(46.6%)*	27.1%	$540,006,000 *(+/-33%)*				
WV	32,509 *(18.0%)*	32,509 *(100%)*	28.8%	$204,923,000 *(+/-35%)*				
TOTALS	180,967 *(100%)*	112,705 *(62.3%)*	100%	$1,326,226,000 *(+/-34%)*	$826,238,798 (+/-42%)	$1,183,278,641 (+/-42%)	$357,039,843 (+/-42%)	**$239,216,695** (+/-42%)

[a] U.S. EPA, unpublished estimates from methods described in Herlihy et al. (2000).
[b] U.S. Dept. of the Interior and U.S. Dept. of Commerce, 1998a-d.
[c] Loomis, 2006.

Table 2 provides the data and steps used to derive regional estimates of actual, potential, and foregone expenditures from sportfishing in mountain streams. For each state in the study region (column 1), we report the total wadeable stream length and the stream length just within the central Appalachians (columns 2 and 3), the contribution of each state's stream length to the total for the study region (column 4), the total 1996 freshwater fishing expenditures estimated in each state, (column 5), and the estimate of total expenditures across the study area, based on the 62.3% of the four-state total stream resource that occurs there (column 6). We attribute this amount to the wadeable streams supporting sportfish species (44.2% of the study region). Because the EMAP survey was not designed for subregional assessments, we cannot estimate each state's individual contribution from EMAP estimates of sportfish presence. From this regional estimate, column 7 shows the projected expenditures across the study region if sportfish had been found in all potentially suitable streams (63.3% [44.2% + 19.1%] of the study region). Column 8 shows the total estimated lost recreational expenditures that we attributed to the proportion of the region with degraded stream conditions (column 7 minus column 6). Finally, column 9 reduces the amount in column 8 to 67% to account for diminished demand in response to increased supply (Loomis, 2006).

We suggest a rough error margin of 42% for all of the region-wide expenditure estimates (columns 6 – 9). Considering the error margins from column 5 and the contributions of each state to the four-state total (column 2), the error margin around the four-state expenditure estimate is approximately 34 percent. This error estimate is in addition to the 8% margin of error associated with the estimate of 19.1% regional stream degradation that was derived from applying the modeling results to the survey data.

5. Discussion

The absence of sportfish species from approximately 30% of potentially suitable wadeable streams in the central Appalachian study area represents a significant loss of recreational opportunity. Our estimate of US$139 – 340 million (in 1996 dollars) in lost value translates to US$6,445 – 15,780 per stream km. These estimates are high compared to those of Knowler et al. (2003), who estimated a minimum value of US$939 – 4,977 per km of stream length for salmon. However, their results derived from a bioeconomic model rather than reported recreational expenditures and are thus not directly comparable. An estimated US$826 million were expended on freshwater recreational fishing across the study region in 1996 alone. Given survey findings that approximately 30% of the region's potentially suitable stream resource contained no sportfish, our analysis suggests that an additional US$240 million (in 1996 dollars) could be available each year to local economies across the region through the restoration of this ecosystem service. This economic value would accompany additional benefits such as improved water quality for other wildlife and for human consumption.

Our finding of significantly higher BIBI (benthic IBI) values in stream segments containing sportfish, as compared to those segments where no sportfish were recorded, bolsters our presumed linkage between sportfish and stream health. While the BIBI is not a surrogate for fish community composition (Griffith et al., 2005), it is a reasonable indicator of the condition of sportfish streams since benthic macroinvertebrates provide an important food source for sportfish. A fish IBI was developed for this region (McCormick et al., 2001), but was not calculated for many segments where sportfish were observed due to insufficient counts of other species making up the index. The fish IBI is poorly suited for the purposes of this study, as it assigns the highest scores to large, warm streams that may promote high fish diversity but do not support sportfish (Rashleigh et al., 2005). Furthermore, the presence of non-native species lowers the score, while the recreational service of wadeable streams includes the provision of both native and non-native sportfish as a benefit.

Several assumptions and qualifiers accompany our study methods and interpretation of results related to the sportfish models. Modeled habitat suitability for the four sportfish species, based on observed occurrences in the training sites, may have over- or underestimated suitable habitat across the study region. Stocking and relocation of both native and introduced species, and the spatial heterogeneity of these efforts, affect sportfish occurrence, yet these efforts are not readily quantifiable at the scale of our analysis. In our results, seven sites where sportfish were found were predicted to not support any sportfish species. We did not filter for canals, ditches, pipelines, or dams (Flebbe et al., 2006); the presence of these structures can restrict sportfish presence in isolated but otherwise suitable stream reaches. We also did not consider species interactions, which can exclude fishes from suitable habitat (Milner et al., 2003). Additional geophysical variables, such as geology and alkalinity, may be important in refining the species range models. For example, brook trout may be intolerant of alkaline streams and areas of Pottsville sandstone in this region (Kocovsky and Carline, 2006). We developed our approach for multi-state assessment units; we do not recommend it to project habitat suitability at fine scales.

The EMAP survey was designed for regional estimates, so we did not calculate the proportion of each state expected to lack sportfish from potentially suitable streams. As a result, we could not estimate recreational expenditures at the sub-state level. All recorded expenditures occurred in the states of interest, even if made by out-of-state travelers. However, we have likely overestimated the portion of lost expenditures attributed to equipment, as this is a capital investment that does not increase proportionately with the time spent in recreational activity. A more conservative estimate could omit equipment expenditures, although items such as clothing and lures purchased in-state by tourists may represent important local revenue. Even so, we assumed but do not know that fishing expenditures are directly related to the amount of streams with sportfish present.

Two factors may have influenced the interpretation of results related to our economic approach. First, we attributed all reported freshwater fishing expenditures (excluding the Great Lakes) to wadeable mountain streams, although we recognized that a portion of the expenditures may have been associated with fishing in lakes and rivers. While this attribution could have inflated expenditures associated with wadeable mountain streams, it could also have underestimated them if freshwater fishing in a state was conducted disproportionately in mountain streams. It should be noted that first- to third-order streams represent 89% of the flowing surface water in the study area (U.S. EPA, 2000), while lakes and reservoirs are relatively scarce (T. Olsen, U.S. EPA National Lakes Assessment, personal communication, 2008). Secondly, we accounted for potential diminished demand with increased supply by applying a 0.67 multiplier to our estimate of annual net value lost from regional sportfish stream degradation (Loomis, 2006). We could have used a more conservative multiplier such as 0.50, which would result in an annual loss estimate of US$179 million (+/- US$75 million) in 1996 dollars, or US$4,810 - $11,776 per stream km.

Despite these assumptions, we believe that our general approach is a first, reasonable approximation of the economic loss associated with the degradation of wadeable sportfish streams across a multistate assessment region. This approach is robust and transferrable, and requires only: 1) classifying ecological condition and quantifying the current provision of nature-based recreational opportunity using regional, probability-based ecosystem monitoring data; 2) projecting potential provision of the ecosystem service across the region by fitting a predictive habitat model to the ecological sampling data; and 3) estimating realized and lost value with regional, probability-based recreational expenditure data. A predictive habitat model may take many forms (e.g., Guisan and Zimmerman, 2000); application of recreational expenditure data as shown in Table 2 may vary under different assumptions. Probability-based, multi-state ecosystem monitoring and recreational user surveys provide the unbiased, large-scale consistent data that are the core of this approach. We propose our method to estimate the potential economic value of degraded wadeable sportfish streams in additional regions where high-quality probabilistic monitoring data exist, such as the 12-state EMAP-West assessment region for the years 2000-2004 and EPA's 2006 National Wadeable Streams Assessment (U.S. EPA, 2006). With comparable aquatic monitoring conducted over time across all 50 U.S. states, we will be able to track regional trends in the current and potential value of ecosystem services that are readily interpreted from sampled metrics.

A full cost-benefit analysis would include improvements in multiple ecosystem services as well as costs associated with watershed- and reach-level resource protection and restoration. Recreation is only one of several important provisioning services associated with healthy streams; others include safe drinking water and wild food species for subsistence diets. In addition, healthy streams play a role in supporting services such as nutrient cycling and primary production. Therefore, our analysis estimated only a portion of what is lost to the region from the degradation of freshwater streams. However, as recreational fishing is readily monetized and highly valued in the market economy, it serves as a fairly straightforward illustration of the degree to which ecosystem services can affect the well-being of the regional population. Ideally, opportunity costs associated with the loss of sportfishing and other services should be weighed against the costs required for stream restoration. Costs vary widely depending on local circumstances (Holmes et al., 2004). Bernhardt et al. (2005) estimated that median project costs range from US$15,000 for riparian management to US$812,000 for land acquisition. Some stressors, such as global warming and acidification, are difficult to remediate (Heft, 2006), so complete restoration may not be possible for all degraded streams. Costs for additional stocking in restored streams may need to be considered as well.

Tourism, recreation, and amenity-based development can increase the number of people that visit a region and provide additional economic stability. Based on the lost opportunity costs from a km of degraded stream mile in our central Appalachian study area, we estimate an approximately 1 year return on investment when stream restoration costs are between $1.96/ft - $4.81/ft. In the Little Tennessee River in the southern Appalachian Mountains, restoration costs of $5.72/ft were considered to be economically feasible (Holmes et al., 2004). Thus, even for a mix of restoration activities used in this watershed, it would only require 2-3 years amortization of the restoration costs for a positive return on investment. The restoration of streams within the region would provide jobs and improve the aesthetics and recreational fishing benefits. Potentially, the expertise in stream restoration within the region could develop into a new industry with an exportable industrial technology and provide additional economic viability.

Acknowledgements

We are grateful to Tony Olsen, Dave Peck, and Tom Kincaid of the U.S. EPA, and to Alan Herlihy of Oregon State University, for their assistance with data from the EMAP mid-Atlantic highlands wadeable streams assessment. We would like to thank Craig Barber, Jim Lazorchak, Marisa Mazzotta, and Alex Macpherson from the U.S. EPA and Lisa Wainger of the University of Maryland for consultation and manuscript review. We are also grateful for the suggestions of four anonymous reviewers. This work was funded by the U.S. Environmental Protection Agency. The National Health and Environmental Effects Research Laboratory approved this document for publication. This approval does not signify that the contents reflect the views of the Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

References

Argent, D.G., Bishop, J.A., Stauffer, J.R. Jr., Carline, R.F., and Myers, W.L. 2003. Predicting freshwater fish distributions using landscape-level variables. *Fisheries Research* 60: 17-32.

Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B., and Sudduth, E. 2005. Synthesizing U.S. river restoration efforts. *Science* 308: 636-7.

Butcher, J.T., Stewart, P.M., and Simon, T.P. 2003. A benthic community index for streams in the northern lakes and forests ecoregion. *Ecological Indicators* 3: 181-93.

Cleveland, C.J., Betke, M., Federico, P., Frank, J.D., Hallam, T.G., Horn, J., Lopez, J.D. Jr., McCracken, G.F., Medellin, R.A., Moreno-Valdez, A., Sansone, C.G., Westbrook, J.K., and Kunz, T.H. 2006. Economic value of the pest-control service provided by Brazilian free-tailed bats in south-central Texas. *Frontiers in Ecology and the Environment* 4: 238-43.

Costanza, R., Pérez-Maqueo, O., Martinez, M.L., Sutton, P., Anderson, S.J., and Mulder, K. 2008. The value of coastal wetlands for hurricane protection. *Ambio* 37: 241-8.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., and van den Belt, M. 1998. The value of the world's ecosystem services and natural capital. *Nature* 387: 253-60.

Daily, G. (ed.). 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, DC: Island Press, 412 pp.

Daily, G.C., and Ellison, K. 2002. *The New Economy of Nature: The Quest to Make Conservation Profitable*. Washington, DC: Island Press, 260 pp.

Davis, W., and Scott, J. (eds.). 2000. *Mid-Atlantic Highlands Streams Assessment: Technical Support Document*. EPA-903-B-00-004. Fort Meade, MD: U.S. Environmental Protection Agency, 96 pp.

Dodds, W.K., Bouska, W.W., Eitzmann, J.L., Pilger, T.J., Pitts, K.L., Riley, A.J., Schloesser, J.T., and Thornbrugh, D.J. 2009. Eutrophication of U.S. freshwaters: analysis of potential economic damages. *Environmental Science and Technology* 43: 12-19.

Elith, J., Leathwick, J.R., and Hastie, T. 2008. A working guide to boosted regression trees. *Journal of Animal Ecology* 77: 802-813.

Flebbe, P.A., Roghair L.D., and Bruggink, J.L. 2006. Spatial modeling to project southern Appalachian trout distribution in a warmer climate. *Transactions of the American Fisheries Society* 135: 1371-82.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., and Snyder, P.K. 2005. Global consequences of land use. *Science* 309: 570-4.

Griffith, M.B., Hill, B.H., McCormick, F.H., Kauffman, P.R., Herlihy, A.T., and Selle, A.R. 2005. Comparative application of indices of biotic integrity based on periphtyon, macroinvertebrates, and fish to southern rocky mountain streams. *Ecological Indicators* 5: 117-36.

Guisan, A., and Zimmermann, N.E. 2000. Predictive habitat distribution models in ecology. *Ecological Modelling* 135: 147-186.

Hall, C.A.S., and Klitgaard, K.A. 2006. The need for a new, biophysical-based paradigm in economics for the second half of the age of oil. *International Journal of Transdisciplinary Research* 1: 4-22.

Heft, A.A. (ed.). 2006. *Maryland Brook Trout Fisheries Management Plan*. Annapolis, MD: Maryland Department of Natural Resources Fisheries Service, Inland Fisheries Management Division, 129 pp.

Herlihy, A.T., Larsen, D.P., Paulsen, S.G., Urquhart, N.S., and Rosenbaum, B.J. 2000. Designing a spatially balanced, randomized site selection process for regional stream surveys: the EMAP mid-Atlantic pilot study. *Environmental Monitoring and Assessment* 63: 95-113.

Holmes, T.P., Bergstrom, J.C., Huszar, E., Kask, S.B., and Orr III, F. 2004. Contingent valuation, net marginal benefits, and the scale of riparian ecosystem restoration. *Ecological Economics* 49: 19-30.

Horn, C.R., and Grayman, W.M. 1993. Water-quality modeling with EPA reach file system. *Journal of Water Resources Planning and Management* 119: 262-74.

Kareiva, P., and Marvier, M. 2007. Conservation for the people. *Scientific American* 297: 50-7.

Knowler, D.J., MacGregor, B.W., Bradford, M.J., and Peterman, R.M. 2003. Valuing freshwater salmon habitat on the west coast of Canada. *Journal of Environmental Management* 69: 261-273.

Kocovsky, P. M., and Carline, R.F. 2006. Influence of landscape-scale factors in limiting brook trout populations in Pennsylvania streams. *Transactions of the American Fisheries Society* 135: 76-88.

Kumar, M., and Kumar, P. 2008. Valuation of the ecosystem services: a psycho-cultural perspective. *Ecological Economics* 64: 808-19.

Lazorchak, J.M., Klemm, D.J., and Peck, D.V. 1998. *Environmental Monitoring and Assessment Program – Surface Waters: Field Operations and Methods for Measuring the Ecological Condition of Wadeable Streams*. EPA/620/R-94/004F. Washington, DC: U.S. Environmental Protection Agency, 211 pp.

Liu, C, Berry, P.M., Dawson, T.P., and Pearson, G.P. 2005. Selecting thresholds of occurrence in the prediction of species distributions. *Ecography* 28: 385–393

Loomis, J. 2006. Use of survey data to estimate economic value and regional economic effects of fishery improvements. *North American Journal of Fisheries Management* 26: 301-7.

Loomis, J., Kent, P., Strange, L., Fausch, K., and Covich, A. 2000. Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. *Ecological Economics* 33: 103-117.

Losey, J.E., and Vaughn, M. 2006. The economic value of ecological services provided by insects. *Bioscience* 56: 311-23.

Ludwig, D., 2000. Limitations of economic valuation of ecosystems. *Ecosystems* 3: 31-5.

Maxted, J.R., Barbour, M.T., Gerritsen, J., Poretti, V., Primrose, N., Silvia, A., Penrose, D., and Renfrow, R. 2000. Assessment framework for mid-Atlantic coastal plain streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 19: 128-44.

McCormick, F.H., Hughes, R.M., Kaufmann, P.R., Peck, D.V., Stoddard, J.L., and Herlihy, A.T. 2001. Development of an index of biotic integrity for the mid-Atlantic highlands region. *Transactions of the American Fisheries Society* 130: 857-77.

Milner, N.J., Elliott, J.M., Armstrong, J.D., Gardiner, R., Welton, J.S., and Ladle, M. 2003. The natural control of salmon and trout populations in streams. *Fisheries Research* 62: 111-25.

National Research Council. 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington, DC: National Academies Press, 277 pp.

Nunez, D., Nahuelhual, L., and Oyarzun, C. 2006. Forests and water: the value of native temperate forests in supplying water for human consumption. *Ecological Economics* 58: 606-16.

Ojeda, M.I., Mayer, A.S., and Solomon, B.D. 2008. Economic valuation of environmental services sustained by water flows in the Yaqui River Delta. *Ecological Economics* 65: 155-166.

Pearce, D.W. 2001. The economic value of forest ecosystems. *Ecosystem Health* 7: 284-96.

Rashleigh, B., Parmar, R., Johnston, J.M., and Barber, M.C. 2005. Predictive habitat models for the occurrence of stream fishes in the Mid-Atlantic Highlands. *North American Journal of Fisheries Management* 25: 1353-66.

Ricketts, T.H., Daily, G.C., Ehrlich, P.R., and Michener, C.D. 2004. Economic value of tropical forest to coffee production. *Proceedings of the National Academy of Sciences* 101: 12579-82.

Ridgeway, G. 2010. Generalized boosted regression models. Documentation on the R package "gbm", version 1.6-3.1. www.i-pensieri.com/gregr/gbm.shtml.

Robinson, J.P., and Godbey, G. 1999. *Time for Life: The Surprising Ways Americans Use Their Time*. University Park, PA: The Pennsylvania State University Press, 367 pp.

Salzman, J. 2006. A field of green? The past and future of ecosystem services. *Journal of Land Use and Environmental Law* 21: 133-51.

Schmitt, C.J., Lemly, A.D., and Winger, P.V. 1993. *Habitat suitability index model for brook trout in streams of the Southern Blue Ridge Province: surrogate variables, model evaluation, and suggested improvements*. Biological Report 18. Washington, DC: U.S. Fish and Wildlife Service, Department of the Interior.

Shafer, E.L., Carline, R., Guldin, R.W., and Cordell, K. 1993. Economic amenity values of wildlife: six case studies in Pennsylvania. *Environmental Management* 17: 669-82.

Smith, V.K., and Huang, J.-C. 1995. Can markets value air quality? A meta-analysis of hedonic property value models. *Journal of Political Economy* 103: 209-27.

Stevens, D.L., Jr., and Olsen, A.R. 2004. Spatially-balanced sampling of natural resources. *Journal of the American Statistical Association* 99: 262-78.

Stoll, J., Ditton, R., and Eubanks, T. 2006. Platte River birding and the spring migration: humans, value, and unique ecological resources. *Human Dimensions of Wildlife* 11: 241-54.

The Nature Conservancy. 2009. Economic benefits of land conservation: A case for Florida Forever. [www.fnps.org/chapters/tarflower/pdfs/economic_benefits_of_land_conservation.pdf].

U.S. Dept. of Commerce. 2007. *Industry Economic Accounts - Interactive Tables: GDP by industry, 1947 - 2006*. www.bea.gov/industry. Washington, DC: Bureau of Economic Analysis.

U.S. Dept. of the Interior, and U.S. Dept. of Commerce. 2002. *2001 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*. Washington, DC: U.S. Fish and Wildlife Service and Bureau of the Census.

U.S. Dept. of the Interior, and U.S. Dept. of Commerce. 1998. *1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*. FHW/96-MD, PA, VA, WV. Washington, DC: U.S. Fish and Wildlife Service and Bureau of the Census. www.census.gov/prod/3/98pubs/mdfhw698.pdf.

[U.S. EPA] U.S. Environmental Protection Agency. 2006. Wadeable *Streams Assessment*. EPA-841-B-06-002. Washington, DC: U.S. Environmental Protection Agency, 98 pp.

[U.S. EPA] U.S. Environmental Protection Agency. 2000. *Mid-Atlantic Highlands Streams Assessment*. EPA-903-R-00-015. Washington, DC: U.S. Environmental Protection Agency, 64 pp.

Walsh, R.G., Loomis, J.B., and Gillman, R.A. 1984. Valuing option, existence, and bequest demands for wilderness. *Land Economics* 60: 14-29.

Weber, M.A. and Stewart, S. 2008. Public values for river restoration options on the Middle Rio Grande. *Restoration Ecology* 17: 762-771.

Wilson, M.A., and Carpenter, S.R. 1999. Economic valuation of freshwater ecosystem services in the United States: 1971 – 1997. *Ecological Applications* 9: 772-83.