perspective paper

BIODIVERSITY

John C. Whitehead
Paul E. Chambers

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John C. Whitehead
Department of Economics
Appalachian State University
Boone, NC 28607

Paul E. Chambers
Department of Economics and Finance
University of Central Missouri
Warrensburg, MO 64093

I. Introduction

This perspective paper reviews the Copenhagen Consensus 2012 Challenge Paper on Ecosystems and Biodiversity (Hussain et al. 2012). The challenge paper addresses the benefits and costs of three programs: increasing agricultural productivity, increasing protected areas and reducing deforestation. The major economic problem addressed by these policies is that biodiversity is a public good. Since its consumption is non-rival and non-excludable, households attempt to free-ride on the payments of others and market allocations result in too little protection of ecosystems and biodiversity. Benefit-cost analysis can be used to identify government policies that would generate positive net benefits for ecosystems and biodiversity reallocation.

While net beneficial policies can be identified, there is a clear mismatch in the incidence of benefits and costs. Households in rich countries tend to benefit while the less rich countries that host the ecosystems and biodiversity tend to bear the costs. In this context, benefit-cost analysis should not be conducted in a vacuum and policies with appropriate management incentives should be considered when conducting the analysis. In the next section of this paper we describe some of these policies, e.g., an increased funding of taxonomic surveys, efficient environmental agreements, asymmetric information, and ecotourism with a focus on volunteer tourism and ecosystem protection designed to avoid fragmentation. Given the numerous policy missteps in this area, we will focus on well-meaning policies that have resulted in unintended consequences. In addition to traditional empirical studies of policies, our analysis will be enriched with insights from common pool resource and public goods experiments from the field and the laboratory. Finally, we will consider the game theoretical aspects of various policies.

The economic benefit of biodiversity and ecosystems is comprised of three components (Nijkamp, Vindigni and Nunes 2008). Market values include medicinal and ecotourism functions of ecosystems. Assuming that prices do not change as a result of policy, these values can be measured as changes in expenditures that result from changes in habitat protection with relatively small error. Use values include the nonmarket value of ecotourism and other recreation activities involving ecosystems. Use values can be measured with revealed preference methods (e.g., travel cost method) and stated preference methods (e.g., contingent valuation method). Non-use values (or passive use values) arise from simply knowing that biodiversity and ecosystems are protected for current and future generations. Non-use values can be estimated with stated preference methods. Use and non-use values are estimated with greater error than market values since non-market valuation methods are used. Hussain et al. (2012) sums market values, use values and nonuse values as if they are equivalently determined dollar values. In the third section of this paper we survey the meta-analysis and benefit transfer literature to determine the reliability of the benefit estimation approach presented in the challenge paper.

The fourth section of the paper will present an analysis of data from the World Values Survey in order to gain insights about the value of biodiversity and ecosystems in a large number of countries. The 2005-2008 World Values Survey wave contains questions concerning willingness to pay for local and global environmental problems. Analysis of these data offers important insights about benefit transfer of biodiversity values. For example, respondents are willing to give up part of their income for environmental problems as their income rises. Holding income and other socioeconomic variables constant, there is considerable heterogeneity in attitudes about willingness to pay across countries.

The next three sections of the paper point to one conclusion; the benefits of ecosystems and biodiversity are difficult to measure and, when measured successfully, subject to a high degree of statistical and other uncertainties. In contrast, the benefit estimates used by Hussain et al. (2012) do not address these uncertainties, relying on point estimates of the present value of benefits for the benefit-cost analysis. In the fifth section of this paper we conduct sensitivity analysis over the estimates of benefits and costs presented by Hussain et al. (2012) using Monte Carlo simulation over ranges of uncertainty. We conduct a series of simulations in order to determine the level of uncertainty at which the benefit-cost ratios are statistically unreliable.

2. Insights from the Economics of Biodiversity

Uncertainty is considered to be an intrinsic part of biodiversity. The uncertainty of the net benefits of biodiversity takes on numerous forms. Biodiversity is difficult to define, measure and quantify. With respect to the definition of obscenity in the case, Jacobellis v. Ohio, 378 U.S. 184 (1964), United States Supreme Court Justice Potter Stewart famously wrote, “I shall not today attempt further to define the kinds of material… [b]ut I know it when I see it.” Similarly, the measurement of biodiversity is not easily defined and it may not even pass Justice Stewart’s criterion given that it may not be easily recognizable. While there is no consensus regarding how biodiversity should be measured (Delong 1996), there are several theoretical measurements of biodiversity including the widely used Simpson and Shannon Weaver indices (Magurran 2004) and distance functions (Weitzman 1992, 1993, 1998; Solow et al. 1993 and Weikard 2002). Weitzman (1995) argues that the value of theoretical measures is not in their practical application but as “a paradigm for guiding and informing conservation decisions, even if the model must be at a high level of abstraction.” In terms of practical measures, taxonomic inventories or species censuses are valuable and Wheeler (1995) described taxonomic inventories as “our only insurance against bio-ignorance.”

Commonly, conservation biologists use species richness, the number of species present in a community or a sample, when considering species diversity. Magurran (2004) describes species richness as the simplest way to describe community diversity. However, the global biodiversity model, IMAGE-GLOBIO3, used by Hussain et al. (2012) does not focus on species richness but on species abundance. Although species abundance is important, it may occur with limited species richness. Contrasting with the Biodiversity Intactness Index which gives greater weight to areas with higher species richness, the Mean Species Abundance (MSA) metric, every hectare is given equal weight (Alkemade et al. 2009). Spatial heterogeneity and the existence of biodiversity hotspots imply that there are significant benefits to parcel selection based on sampling. Biodiversity hotspots may take three forms: overall species richness, threat and endemism with the overall species richness hotspots not necessarily being congruent with the threatened and endemic hotspots (Orme et al. 2005). Naidoo et al. (2006) found significant cost reductions when the spatial heterogeneity of diversity was considered, suggesting that the use of equal weights results in inefficiencies.

In addition to a greater consideration of species richness, parcel selection should consider the uncertain effects of fragmentation. If fragmentation is present, biodiversity may be compromised (Fahrig 2003). As Fahrig notes, it is important to distinguish between habitat loss and habitat fragmentation. Habitat loss has negative effects on biodiversity; however, fragmentation may have uncertain effects and disproportionally affect adversely certain species (Ewers and Didham 2006).

Hussain et al.'s (2012) policy prescription of extending protected areas to prevent the loss of biodiversity raises the question: Is environmental diversity a substitute for biodiversity? Environmental diversity is the establishment of reserves for the protection of biodiversity. However, Araújo et al (2001), find that environmental diversity areas are poor surrogates for terrestrial vertebrates. Oliver et al. (2004) find support for environmental surrogates but with a caveat of additional sampling of geographic space or range occupied by each land system. If environmental diversity is an imperfect substitute for the maintenance of biodiversity, we recommend expenditures on taxonomic surveys. Since certain taxa (especially invertebrates) are poorly documented (Magurran 2004), there exists a need for greater sampling. E.O. Wilson (1991) has argued that "systematics or the study of biological diversity and its origins has one of the highest benefits to costs ratios of all scientific disciplines." Unfortunately, the collection of biodiversity data is both time consuming and expensive (Magurran et al. 2010). Although ecotourism has not been the panacea it was once considered (Bookbinder et al. 1998), Brightsmith et al. (2008) find that ecotourism, in the form of volunteer tourism, may be a source of funding and labor for conservation research projects.

There are many factors that make the cost of reserve expansion difficult to project accurately (James et al. 2001). We will focus on two factors, the enforcement of property rights and the existence of potential strategic behavior that may lead to greater uncertainty and a systematic underestimation of the costs of conservation. International environmental agreements, such as debt for nature swaps, are difficult to enforce given that the resources may be owned by a sovereign nation (see Chambers, Jensen and Whitehead 1996; Deacon and Murphy 1997). Chambers and Jensen (2002) show that environmental aid may actually increase environmental degradation if a recipient nation acts in a strategic manner given the presence of asymmetric information. Domestic environmental purchases may be plagued by costly strategic behavior in that the landowner may engage in anticipatory investment (development) to affect the landowner's bargaining position which in turn would adversely affect the conservation organization's costs (see Richer and Stranlund 1997; Stranlund (1999).

Even if the recipient sovereign nation is compliant, the conservation efforts may be ineffective at the local level. Governments may create a national park or reserve, but fail to provide any real protection due to lack of funding or lack of community support. These areas are referred to as "paper parks." Bruner et al. (2001) found that 57% of the tropical parks in their sample had net clearing since their establishment. They found that even modest increases in funding would directly increase the ability of parks to protect tropical biodiversity. In 1990, The Nature Conservancy began an initiative known as Parks in Peril in order to identify the most threatened natural areas and work to convert them from paper parks to fully protected areas. Partnering with USAID and local conservation organizations across Latin America and the Caribbean, Parks in Peril protected 45 parks and reserves covering 44.8 million acres between 1990 and 2007. Their approach involved empowering local residents, developing long term funding strategies, and developing systems of municipal parks for biodiversity conservation. It is not clear if similar provisions are included in the policies described by Hussain et al. (2012).

What is the best practice to avoid paper parks? As Ostrom (1990) and Poteeten, Janssen and Ostrom (2010) have shown, there is no single, universal solution to the institutional fragilities that arise with the commons and collective action problems such as the protection of biodiversity. Successful environmental agreements are still evolving with approaches based on field and laboratory experiments. Cárdenas (2004) and Cárdenas and Ostrom (2004) provide insights regarding the effects of social norms, internal versus external regulation which may be useful in the development of more efficient management of common pool resources. However, Ostrom (1990) states "that 'getting the institutional right' is a difficult, time-consuming, conflict-invoking process."

3. Insights from the Benefits Transfer Literature

Meta-analysis is a general term for any methodology that empirically summarizes a literature from several studies. It requires the collection of a large number of studies related to the policy situation. In the case of environmental benefit transfer, a data set is constructed with benefit estimates gathered from several studies with willingness to pay as the dependent variable and characteristics of the individual studies serve as the independent variables. Regression models are developed which are used to relate benefit estimates to the characteristics of the study and policy context. These regression models are used as benefit function transfer models where the characteristics from the case study are inserted and environmental benefits related to the case study are developed.

Benefit transfer using meta-analysis has three advantages (Shrestha and Loomis, 2001). First, by employing a large number of studies, the willingness to pay estimates will be more rigorous. Second, methodological differences can be controlled. For example, meta-analysis may be used to control for differences in functional form across studies (Smith and Kaoru, 1990). Third, differences between the study site and the policy site can be better controlled. However, meta-analysis suffers from (1) reporting errors and omissions in original studies, (2) inconsistent definitions of environmental commodities and values, and (3) large random errors. These drawbacks limit the ability to use meta-analysis as a benefit transfer method.

Several early meta-analyses showed the importance of study characteristics (i.e., functional form, publication year) when explaining variations in benefit estimates. Smith and Kaoru (1990) conduct a meta-analysis of willingness to pay estimates derived from travel cost recreation demand models. Smith and Huang (1993) conduct a meta-analysis of air quality benefits derived from hedonic property value models. Loomis and White (1996) conduct a meta-analysis of willingness to pay estimates from studies of rare and endangered species. Each of these studies finds that study characteristics influence benefit estimates.

Other studies have pooled benefit estimates from different non-market valuation methodologies. Walsh, Johnson, and McKean (1992) conduct a meta-analysis of outdoor recreation value estimates from travel cost recreation demand and contingent valuation studies. Woodward and Wui (2001) conduct a meta-analysis of willingness to pay estimates from studies of wetland values using travel cost, contingent valuation and other methods. Carson, et al. (1996) conduct a meta-analysis of studies that employ the contingent valuation and hedonic property value, travel cost, and averting behavior methods. In each of these, estimates of willingness to pay from the contingent valuation method are lower than those from the revealed preference methods. These early studies suggest that non-market valuation methodology is an important independent variable for meta-analyses.

More recently, a number of meta-analyses have appeared in the literature for biodiversity. Following previous research, each study includes a large number of independent variables, including study characteristics. Jacobsen and Hanley (2009) consider contingent valuation studies of biodiversity in order to focus on non-use values and find that the income elasticity of willingness to pay across countries is 0.38. Their meta-analytic model includes variables that control for study origin, focus of the study (habitat vs. species), payment unit, income unit, payment interval, payment vehicle, valuation question format and time of the survey. They find that a few of these methodological variables are statistically significant.

Barrio and Loureiro (2010) estimate a meta-analysis model with contingent valuation studies of forest values. They find that a number of study characteristics, forest characteristics, site characteristics and socioeconomic characteristics help explain the variation in forest values. Studies that include forests that generate biodiversity values (i.e., “management of flora and fauna”) have no statistical effect on the overall values. Recreation opportunities increase forest economic values. In other words, biodiversity is a relatively low component of forest values. The amount of variation in the dependent variable explained by variation in the independent variables ranges (i.e., model R^2) from 83% to 91%.

Ojea and Loureiro (2011) estimate a meta-analysis model with contingent valuation studies of biodiversity with a focus on the issue of scope (i.e., “more is better”). They find that scope effects exist when it is measured absolutely (instead of relatively), economic values tend to be lower if the benefits are mostly nonuse values, that more recent studies generate lower values and that methodology is important in explaining economic values.

A few studies have assessed the accuracy of benefit transfers with meta-analyses. Using an update to the meta-analytic data from Walsh, Johnson, and McKean (1992), Rosenberger and Loomis (2000) find that benefit transfer errors range from 54% to 71%. Shrestha and Loomis (2001) conduct out-of-sample benefit transfer tests using results from the data from Rosenberger and Loomis (2000). In contrast to other studies reviewed here but similar to Hussain et al. (2012) their R^2 statistic is relatively low, 0.26. They use U.S. studies to forecast benefits for international policy sites that are not included in the data. They find that the average prediction error ranges from 24% to 30%.

Lindhjem and Navrud (2007) consider the accuracy of meta-analysis benefit transfer with particular attention to the role of model specification for relatively homogeneous Scandanavian forests. Using regression models of various functional forms, the model R^2 statistic ranges from 86% to 89%. Even with reliable econometric models the authors find that out-of-sample benefit transfer errors range from 21% to 51% when using median willingness to pay and 62% to 266% when using mean willingness to pay. They conclude that meta-analysis benefit transfer may not improve accuracy compared to simpler benefit transfer techniques.

This review raises several concerns about the Hussain et al. (2012) meta-analysis. First, Hussain et al. (2012) do not include variables that identify the type of benefits (market, use, nonuse). This could lead to an "apples vs. oranges" problem where one type of benefit is estimated and transferred to a policy situation that would generate another type of benefit. Second, benefit transfer errors from meta-analyses have ranged from 24% to 71% for recreation studies with relatively low R^2 statistics and 62% to 266% for forest studies (that use mean willingness to pay) with relatively high R^2 statistics. This is little reason to believe that the benefit transfers in Hussain et al. (2012) will be 100% accurate. Third, Hussain et al. (2012) do not include variables that identify ability to pay, study characteristics and study methodology. If omitted variable bias is present in these models it could cause the coefficient estimates to be biased. Bias in the coefficient estimates can lead to additional inaccuracies in benefit transfers.

4. Insights from the World Values Survey

We use the most recent wave of data from the World Values Survey from 2005 to 2007 (WVS, 2009). This wave also includes responses from 2008, but results from 2008 seem counter to the results from 2005-2007, perhaps due to the effects of the global recession. The countries included in each year of the WVS vary (see Appendix), but include both developed and developing countries. After dropping cases with missing values on key variables, our sample includes 45,435 observations from 41 countries.

Our dependent variables are attitudinal measures of the economic benefits of global environmental quality. Respondents are presented the following questions: "I am now going to read out some statements about the environment. For each one I read out, can you tell me whether you strongly agree, agree, disagree, or strongly disagree." The two statements assessed in this paper are: (1) "I would be willing to give part of my income if I were sure that the money would be used to prevent environmental pollution," and (2) "I would agree to an increase in taxes if the extra money is used to prevent environmental pollution." Sixty-nine percent of the sample strongly agrees or agrees that they would be willing to give up income and sixty-two percent of the sample strongly agrees or agrees that they would be willing to pay higher taxes. The frequency of responses across the two questions differ significantly ($p < 0.001$) which indicates that the method of payment matters and that separate analysis of both questions could lead to additional insights.

In order to relate the attitudinal statements of willingness to pay to ecosystem and biodiversity benefits we consider attitudinal statements about global environmental problems: "Now let's consider environmental problems in the world as a whole. Please tell me how serious you consider each of the following to be for the world as a whole. Is it very serious, somewhat serious, not very serious, or not serious at all?" The three global problems are "global warming or the greenhouse effect," "Loss of plant or animal species or biodiversity," and "Pollution of rivers, lakes, and oceans." Fifty-six percent believes that the loss of biodiversity is a very serious problem, 33 percent feel that it is somewhat serious and only 11 percent feel that it is not very serious or not serious at all. Eighty-nine percent of the sample believes that global warming is very or somewhat serious. Ninety-two percent believes that pollution of rivers, lakes and oceans is a very or somewhat serious problem.

While these three variables are highly correlated, the frequency of responses across the three questions differ significantly ($p < 0.001$) which indicates that the three issues could be treated separately in the empirical model. We recode these variables so that each is equal to one if the respondent felt that the problem is very serious.

Respondents were also asked about their attitudes towards local environmental problems: "I am going to read out a list of environmental problems facing many communities. Please tell me how serious you consider each one to be here in your own community. Is it serious, somewhat serious, not very serious, or not serious at all?" The three issues are "Poor water quality," "Poor air quality," and "Poor sewage and sanitation." Attitudes towards local environmental problems are highly correlated, causing multicollinearity in the empirical models, so we code each "very serious" attitude equal to one and sum these across respondents. We include this as an independent variable to control for the contribution of concern about local, relative to global, environmental problems. The average level of local environmental concern is 1.24.

In addition to local environmental concern, we include several socioeconomic control variables in the empirical models. Fifty percent of the sample is female, the average age is 41 (range is from 15 to 97 years), fifty-four percent of the sample is married and the average number of years schooling is 11 (the range is from 6 to 16 years). The income variable is the income decile that the respondent perceives themselves to inhabit. The average income decile is 4.92. Socioeconomic summaries for each country in the sample are presented in the appendix.

In order to conduct the empirical analysis we recode the willingness to pay attitudinal variables to equal one if the respondent strongly agrees and zero otherwise. We estimate the determinants of strongly agree willingness to pay with logistic regression. Similar results are found with ordered logistic regression; however, we estimate simpler models given the limited scope of this paper. In addition to the above mentioned independent variables we include dummy variables for countries. We constrain similarly sized coefficient estimates and conduct a series of likelihood ratio tests to determine groups of countries that have like parameters. For the income decile variable, we create dummy variables for each decile and constrain coefficients to be equal using the results from likelihood ratio statistics. We also conduct similar tests for year of survey dummy variables.

In both models, respondents are more likely to strongly agree with the willingness to pay statements as their perceptions of the seriousness of local environmental problems increase. Respondents also strongly agree about willingness to pay for each of the global environmental problems. In the income model, none of the global environmental problem coefficients are statistically different from each other according to likelihood ratio statistics. According to the odds ratio statistic, respondents are 1.32, 1.26 and 1.21 times more likely to be willing to give up some of their income if they feel that global warming, biodiversity and water pollution are very serious global problems. In the taxes model, respondents are 1.16, 1.27 and 1.21 times more likely to be willing to pay higher taxes if they feel that global warming, biodiversity and water pollution are serious global problems. The coefficient on biodiversity is not statistically different from water pollution but is statistically greater than global warming according to likelihood ratio statistics. However, none of the differences in the odds ratios are statistically significant.

In terms of the socioeconomic factors, the probability that respondents strongly agree about giving up income increases with age, for those who are married with increasing years of schooling and with income. Respondents who are in the fourth and fifth, sixth and seventh and above income deciles are more (and more) likely to strongly agree about willingness to give up some income.

In terms of the socioeconomic factors, the probability that respondents strongly agree about paying taxes increases for those who are married with increasing years of schooling and with income. Respondents who are in the fourth and fifth, sixth, seventh and eighth and above income deciles are more and more likely to strongly agree about willingness to pay taxes.

In the income model, we find ten separate sets of countries with differing attitudes about willingness to give up income. In the second model, willingness to pay higher taxes, we find nine separate sets of countries with differing attitudes. There are some similarities in groups of countries but the only commonality is Viet Nam (number 39) whose respondents are six and seven times more likely to give up income and pay higher taxes than the baseline group of countries. The next highest odds ratio in the willingness to give up income model is 1.90 and 1.66 in the taxes model. The odds ratios for Viet Nam would be even greater if the omitted country group is that which is least willing to pay.

Finally, in terms of survey year fixed effects, respondents in the 2006 and 2007 surveys are more likely to be willing to give up income, relative to 2005. Respondents in 2006 are less likely to be willing to pay more taxes while respondents in the 2007 surveys are more likely to be willing to pay.

Our analysis of the World Values Survey data suggests two salient points for the assessment of Hussain et al. (2012). First, if the correlation between concern about environmental problems and willingness to pay is causal, there is worldwide support for protection of biodiversity. However, these attitudinal statements do not provide evidence of the magnitude of willingness to pay. Given that considerable effort has been made by a number of researchers to create meta-analytic databases of biodiversity valuation studies, future effort might usefully be redirected toward the conduct of multicountry contingent valuation surveys of biodiversity (Carson 1998). Second, meta-analytic models, even if adjusted for income, socioeconomic variables and attitudes about biodiversity, are likely to be less than 100% accurate when used for benefit transfers. This is due to country level fixed effects and the temporal nature of attitudes towards biodiversity. The meta-analyses in Hussain et al. (2012) do not incorporate variables of this nature.

5. Sensitivity Analysis

Hussain et al. (2012) find that the benefit-cost ratios for policies that enhance agricultural productivity, protect land and reduce deforestation are 4.4, 0.2 to 1.0 and 3.6 to 9.8, respectively (using a discount rate of 3%). Including climate change co-benefits increases these ratios to 7.5 to 20.5, 0.3 to 1.4 and 7.8 to 31.3. Best case and worst case sensitivity analysis is conducted but this leads to ranges of benefit-cost ratios where each ratio might seem to be equally likely. For example, there is little guidance concerning where in the range of 3.6 to 9.8 the benefit-cost ratio for reduced deforestation is most likely to fall.

Our review of biological and institutional issues, the biodiversity valuation and benefit transfer literature and analysis of the World Values Survey data leads us to question the confidence placed in the benefit-cost ratios. Each of these issues suggests that the actualization of the economic benefit of ecosystem and biodiversity is highly uncertain or measured with considerable statistical error. A limitation of the Hussain et al. (2012) analysis is that only point estimates of the benefits of each of the three policies is considered when calculating benefit-cost ratios.

Further, considering the data used for the benefits analysis also raises the issue of uncertainty. For two of the three sets of data, the median benefit estimate is much lower than the mean indicating that the sample is strongly influenced by outliers (the median and mean for the third data set are not reported). Removal of these outliers could result in much lower present values of benefits for each policy and lower benefit-cost ratios. On the other hand, Hussain et al. (2012) assume a linear trajectory of annual benefits that begins at zero in the initial year. Alternative trajectory assumptions, with larger benefits received in early years, would lead to larger present value of benefit estimates and larger benefit-cost ratios.

Considering these uncertainties, we conduct a limited Monte Carlo sensitivity analysis in order to (1) provide a mean benefit-cost ratio for each policy and (2) assess the sensitivity of the mean benefit-cost ratios to assumptions made in the analysis. For each policy we take 1000 random draws from a normal distribution of the present value of benefits with the coefficient of variation (CV) equal to 0.1, 0.2, 0.3, 0.4 and 0.5.

In addition, the agricultural productivity policy assumes certain costs. One-thousand random draws from a normal distribution with CV equal to 0.05, 0.1, 0.15, 0.2 and 0.25 are used for the agricultural productivity policy, assuming that cost estimates are less uncertain than benefit estimates. Ranges of costs are presented for the other policies. One-thousand random draws are taken from a uniform distribution of the low and high cost estimates for the protected areas and deforestation scenarios.

These random draws are used to construct 1000 benefit-cost ratios for which we report mean and standard deviations. The 1000 benefit-cost ratios are ordered from lowest to highest and the 25 lowest and highest values are trimmed to create a 95% confidence interval. We consider the benefit-cost analysis with carbon values included for the protected areas policy since (1) benefit-cost ratios are clearly below one without carbon values for that policy and (2) benefit-cost ratios with carbon values included are quite large for the other policies.

In Table 4 we present the sensitivity analysis of benefit-cost ratios at a 3% discount rate. Considering the agricultural policy, as the CV rises from 0.1 to 0.5 for the present value of benefits and the CV of the present value of cost rises from 0.05 to 2.5 the standard deviation of the benefit-cost ratio rises from 0.52 to 2.72. The 95% confidence interval rises from 46% to 219% of the benefit-cost ratio. At CVs greater than 0.4 the 95% confidence interval of the benefit-cost ratio falls below one, indicating that the ratio is not statistically different from one. The sensitivity analysis for protected areas including carbon values is trivial, with the 95% confidence interval never being statistically different from one. Considering the deforestation policy, as the CV of the point estimate of present value of benefits rises from 0.1 to 0.5 the mean of the benefit-cost ratio varies between 5.60 and 6.00 and the standard deviation rises from 1.78 to 3.17. The 95% confidence interval rises from 109% to 211% of the benefit-cost ratio. At CVs somewhere between 0.4 and 0.5 the 95% confidence interval of the benefit-cost ratio falls below one, indicating that the ratio is not statistically different from one.

6. Conclusions

Hussain et al. (2012) have presented a wealth of information and are to be commended for the magnitude of their effort. However, in this paper we have raised a number of concerns about the analysis.

Uncertainty is an unavoidable component of the study of biodiversity. This inherent uncertainty is reflected in the estimate that the earth is inhabited by 8.7 million species with a standard error of 1.3 million species. Only 1.3 million of these species are currently named (Mora et al. 2011). Not only is the number of species elusive, but a large portion of the biodiversity benefits these species represent is unknown and benefits may occur with unexpected species. For example, the bark of the Pacific yew (*Taxus brevifolia*), a tree previously considered to be a marginal species that was discarded during logging of old growth forests, is the source of the anti-cancer drug, Taxol (see Polasky and Solow 1995, Chivian 2001). With preserved species, such as the Pacific yew which was once considered to be a trash tree by loggers, there is a positive expected value of the future information that extinction would preclude. This option value, as defined by Conrad (1980), Hanemann (1989) and others, is difficult to measure. However, the discovery of previously unknown species and associated benefits through taxonomic inventories and bioprospecting is a costly undertaking. Finally, accurate acquisition and management costs are clouded by asymmetric information and strategic behavior.

Even without these uncertainties, numerous and restrictive conditions are necessary for a benefit function from a meta-analysis to transfer accurately to a policy site. It is not surprising that few studies generate satisfyingly accurate benefit transfers. These and our other concerns raise questions about over-confidence in the point estimates of the present value of benefits presented in the challenge paper (Hussain et al. 2012). Our attempt to conduct ex-post sensitivity analysis suggests that none of the policies are likely to have benefits greater than costs at the 95% level of confidence when the standard deviation is about 40% of the point estimate of the present value of benefits. However, more extensive sensitivity analyses are needed for each policy, with Monte Carlo simulation conducted, ex-ante, over the large number of explicit and implicit assumptions that led to the point estimate of the present value of benefits.

Table 1. Willingness to pay for environmental improvements				
	Would give part of my income for the environment		Increase in taxes if used to prevent environmental pollution	
	Frequency	Percent	Frequency	Percent
Strongly Agree	7838	18.05	6283	14.47
Agree	22,015	50.68	20,479	47.15
Disagree	10,111	23.28	12,254	28.21
Strongly Agree	3471	7.99	4419	10.17

Table 2. Perceptions of Global Environmental Problems						
	Global warming or the greenhouse effect		Loss of plant or animal species or biodiversity		Pollution of rivers, lakes and oceans	
	Frequency	Percent	Frequency	Percent	Frequency	Percent
Very Serious	25,383	58.44	24,400	56.18	28,870	66.47
Somewhat Serious	13,448	30.96	14,220	32.74	11,074	25.50
Not Very Serious	3,795	8.74	4,055	9.34	2,829	6.51
Not Serious at all	809	1.86	760	1.75	662	1.52

Table 3. Logistic Regression Models (Dependent Variable = 1 if agree or strongly agree); n = 43,435

	Would give part of my income for the environment			Increase in taxes if used to prevent environmental pollution		
	Coeff.	S.E.	p-value	Coeff.	S.E.	p-value
Constant	0.032	0.070	0.6451	-0.362	0.071	<.0001
Local	0.039	0.010	<.0001	0.029	0.0091	0.0018
Global warming (=1 if very serious)	0.28	0.028	<.0001	0.15	0.026	<.0001
Biodiversity (=1 if very serious)	0.23	0.030	<.0001	0.24	0.028	<.0001
Water pollution (=1 if very serious)	0.19	0.031	<.0001	0.19	0.029	<.0001
Income Decile (=1)						
4, 5	0.15	0.028	<.0001	0.11	0.027	<.0001
6	0.23	0.037	<.0001	0.22	0.035	<.0001
7				0.40	0.038	<.0001
7, 8, 9, 10	0.32	0.033	<.0001			
8, 9, 10				0.34	0.037	<.0001
Female	0.038	0.022	0.0845	0.019	0.021	0.3468
Age	0.0020	0.00075	0.0071	-0.0007	0.00070	0.3569
Married	0.050	0.024	0.0355	0.065	0.022	0.0034
Schooling	0.066	0.0041	<.0001	0.038	0.0039	<.0001
Countries (=1)						
12, 28, 36, 38	-1.91	0.043	<.0001			
2, 4, 5, 8, 10, 22, 27, 37	-1.32	0.035	<.0001			
1, 4, 7, 14, 19, 40, 41	-0.98	0.039	<.0001			
20, 23, 26, 30	-0.94	0.045	<.0001			
6, 31	-0.67	0.055	<.0001			
16, 21, 29, 34	-0.43	0.043	<.0001			
13, 17	-0.12	0.058	0.0463			
9, 32	0.64	0.069	<.0001			
39	1.79	0.16	<.0001	1.91	0.11	<.0001
22, 28, 38				-1.05	0.048	<.0001
5, 10, 21, 27, 36, 37, 40				-0.54	0.039	<.0001
2, 4, 8, 19, 20, 23, 26, 31, 41				-0.43	0.037	<.0001
1, 7, 14, 16, 34				-0.20	0.040	<.0001
6, 35				0.15	0.051	0.0037
25, 29, 30				0.12	0.045	0.0066
9, 24				0.50	0.067	<.0001
Year (=1)						
2006, 2007	0.070	0.027	0.009			
2006				-0.14	0.028	<.0001
2007				0.41	0.037	<.0001
Model $\chi 2$	5638 (p < 0.0001)			3583 (p < 0.0001)		

Table 4. Sensitivity Analysis of Benefit-Cost Ratios				
Agricultural Productivity				
CV	B/C	Std Dev	95% Confidence Interval	
0.1	4.39	0.52	3.40	5.41
0.2	4.40	1.03	2.55	6.44
0.3	4.51	1.57	1.80	7.79
0.4	4.70	2.21	0.91	9.86
0.5	4.67	2.72	0.40	10.64
	Protected Areas with Carbon Values			
CV	B/C	Std Dev	95% Confidence Interval	
0.1	0.51	0.24	0.29	1.18
0.2	0.58	0.29	0.27	1.24
0.3	0.57	0.27	0.23	1.29
0.4	0.57	0.28	0.17	1.29
0.5	0.59	0.33	0.13	1.44
	Deforestation			
CV	B/C	Std Dev	95% Confidence Interval	
0.1	5.60	1.78	3.39	9.48
0.2	5.82	2.13	2.77	10.80
0.3	6.00	2.57	1.88	11.77
0.4	5.97	2.93	1.17	12.95
0.5	5.94	3.17	0.25	12.79

Appendix. 2005-2007 World Values Survey Data

Country	Code	Year	Cases	Local	Income	Female	Age	Married	Schooling
1	AD	2005	866	0.79	5.60	0.50	40.13	0.41	12.22
2	AU	2005	1242	1.00	5.26	0.53	49.38	0.59	13.30
3	BF	2007	922	2.52	3.84	0.44	33.95	0.52	8.62
4	BG	2006	613	1.90	4.02	0.53	46.17	0.64	12.52
5	BR	2006	1349	1.20	4.32	0.58	39.58	0.41	9.91
6	CA	2006	1562	0.67	5.70	0.56	47.74	0.48	12.43
7	CH	2007	978	0.72	5.45	0.52	51.85	0.54	12.95
8	CL	2005	800	1.27	4.27	0.54	42.25	0.47	11.33
9	CN	2007	783	0.68	4.33	0.47	42.09	0.81	11.00
10	CSS	2006	934	1.91	4.76	0.49	41.56	0.59	12.20
11	CY	2006	1017	1.95	5.69	0.51	41.59	0.65	12.05
12	DE	2006	1517	0.37	4.49	0.53	50.48	0.57	10.74
13	ET	2007	1301	2.11	5.22	0.48	29.62	0.38	10.43
14	FI	2005	874	0.42	4.33	0.51	47.02	0.47	11.75
15	GH	2007	1003	1.39	4.70	0.46	33.12	0.42	9.39
16	ID	2006	1259	1.47	5.37	0.46	34.85	0.18	12.78
17	IN	2006	964	1.62	4.00	0.36	39.76	0.78	11.04
18	IR	2007	2511	2.11	4.88	0.49	32.50	0.60	11.29
19	IT	2005	553	0.61	4.31	0.49	45.59	0.59	12.40
20	JP	2005	524	0.48	4.69	0.48	49.95	0.73	12.81
21	KR	2005	1193	0.21	4.88	0.50	41.35	0.64	13.43
22	MA	2007	677	2.14	5.02	0.46	34.89	0.52	8.92
23	MD	2006	926	1.82	4.99	0.52	41.86	0.64	11.69
24	ML	2007	805	2.30	4.99	0.49	36.59	0.67	8.77
25	MX	2005	1252	1.22	5.10	0.48	38.12	0.54	11.22
26	MY	2006	1192	0.88	5.88	0.50	31.85	0.50	11.73
27	PL	2005	760	1.43	4.08	0.47	45.16	0.58	11.05
28	RO	2005	1113	1.12	6.12	0.50	47.18	0.71	11.37
29	RW	2007	1212	1.35	3.40	0.50	34.26	0.52	8.39
30	SE	2006	880	0.17	6.13	0.47	47.94	0.48	12.97
31	SI	2005	785	0.54	5.08	0.49	44.41	0.51	12.07
32	TH	2007	1446	0.76	5.59	0.51	45.37	0.69	9.85
33	TR	2007	1115	2.22	3.61	0.47	35.82	0.65	10.53
34	TT	2006	833	0.78	4.74	0.54	42.22	0.39	10.80
35	TW	2006	1174	0.34	4.49	0.49	43.21	0.64	12.47
36	UA	2006	650	1.89	4.49	0.64	41.44	0.58	13.25
37	US	2006	1118	1.06	5.03	0.49	47.99	0.58	11.79
38	UY	2006	751	0.95	4.52	0.56	46.41	0.42	10.35
39	VN	2006	1147	1.18	5.62	0.46	40.20	0.73	10.79
40	ZA	2007	1980	1.59	5.14	0.49	38.83	0.42	11.05
41	ZM	2007	854	1.19	5.64	0.47	29.03	0.32	11.37

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