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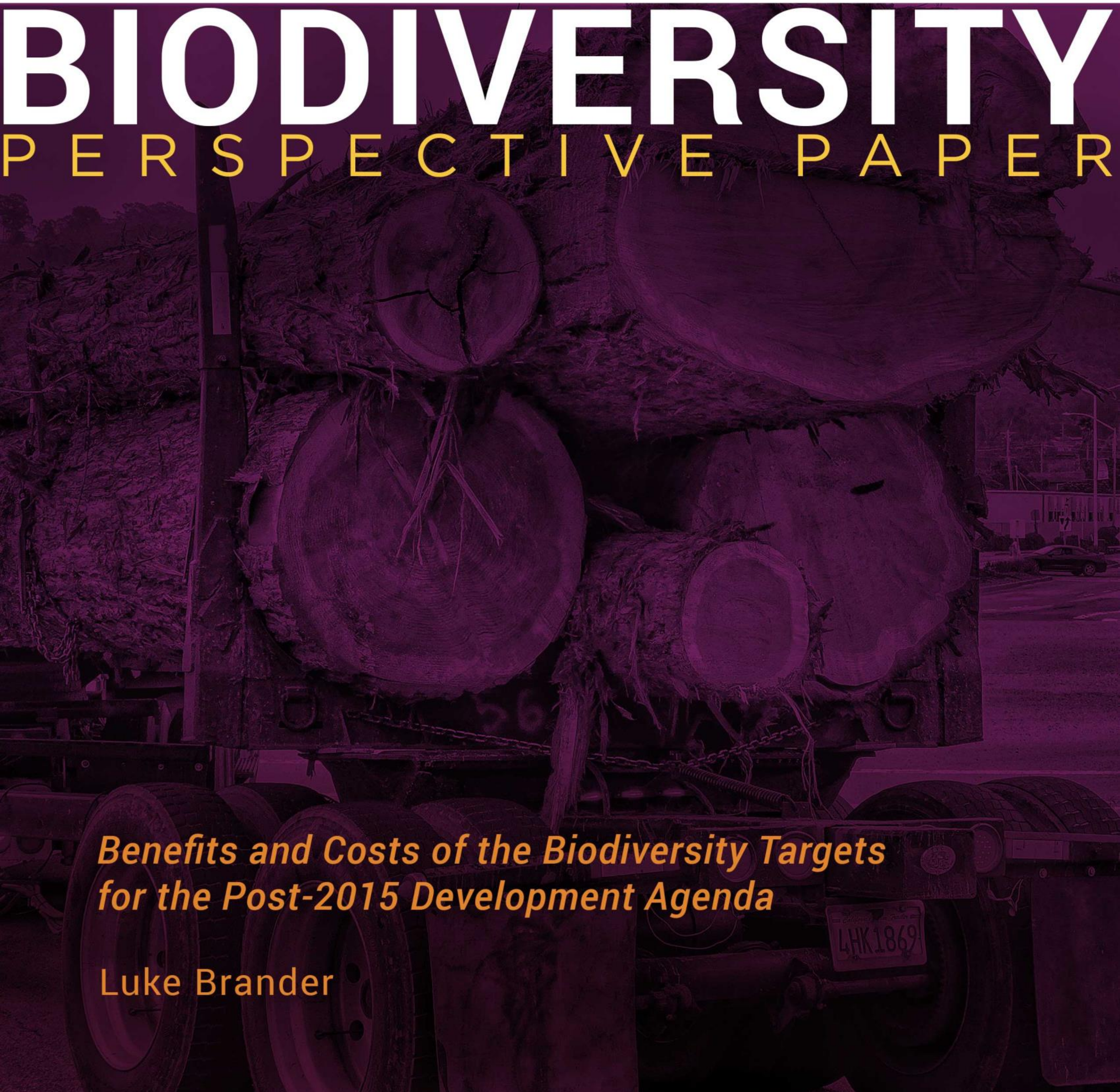

Benefits and Costs of the Biodiversity Targets for the Post-2015 Development Agenda

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Post-2015 Consensus

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Introduction

This perspective paper makes a proposal for improved methodologies for conducting large scale assessments of the costs and benefits of meeting targets for biodiversity and ecosystem change. This proposal may be used for strengthening the analysis presented in the Challenge Paper on biodiversity and deforestation (Markandya, 2014) or for subsequent analyses that aim to examine the costs and benefits of the targets in more detail.

The need for improved methods for large scale (global) assessments of the costs and benefits of conserving biodiversity and reducing the loss of ecosystems is driven by the recognition that currently applied approaches (such as those used in Markandya, 2014) do not produce sufficiently accurate information for use in cost-benefit analyses. An example of this lack of sufficient accuracy is provided by the assessment of the benefits of reducing the rate of wetland loss (Aichi Target 5) in Markandya (2014). For the valuation of benefits, lower and upper bound annual unit (per hectare) values obtained from De Groot et al. (2012) are used. These values are computed by De Groot et al. (2012) as the sum of minimum values for each wetland ecosystem service observed in the primary valuation literature and the sum of all maximum values; and arguably represent the extremes of possible values derived from wetlands.[1] Markandya (2014) applies these lower and upper bound values to the change in global stock of wetlands without adjustment[2] and rightly comments that the "the true values are probably somewhere between the two". The resulting benefit estimates are therefore either very low (and result benefit-cost ratios less than 1 in the subsequent CBA) or very high (and result in benefit-cost ratios greater than 35). This enormous span of results implies that we cannot estimate even the order of magnitude of wetland benefits and as a consequence this information is of limited use for decision making. This problem with using extreme lower and upper bound unit values for ecosystem services is also evident in the analysis of Aichi Target 10 (reducing anthropogenic pressures on coral reefs), for which only a lower bound estimate is used, presumably because the upper bound is implausibly high.

There are several related methodological problems present. Firstly, values for ecosystem services are unlikely to be globally constant across an entire biome and are expected to vary with the characteristics of each ecosystem, the context in which they are supplied (e.g. the availability of substitute ecosystems) and the characteristics of the population that benefit from them (e.g. number of beneficiaries, income, dependence on ecosystem services). Secondly, marginal values for ecosystem services are unlikely to remain constant as the stocks of ecosystems change (i.e. become more scarce). When valuing simultaneous changes in multiple ecosystem sites within a region (i.e. scaling up values), it is not sufficient to estimate the aggregate value of ecosystem services without accounting for the

[1] Summing values for all ecosystem services implicitly assumes that an individual ecosystem provides all services simultaneously (i.e. all ecosystems are utilized to the full and there are no trade-offs in the provision of different ecosystem services). This assumption is unlikely to be valid.

[2] Adjustments can be made to reflect variation in the determinants of economic value, such as the characteristics of the ecosystems providing the services and/or the beneficiaries that use them.

changes that are occurring across the stock of the resource (Brander et al., 2012a). Thirdly, assessing this expected variation in values by using minimum and maximum values in a sensitivity analysis will produce an enormous range of results that does not represent plausible values or provide useful decision rules.

To address these problems we propose to use meta-analytic value transfer methods combined with spatial data on biophysical and socio-economic determinants of ecosystem service values to produce spatially variable and more accurate estimates of benefits (Brander et al., 2012a). This method is applied to re-estimate the benefits of meeting Aichi Target 5 for reducing the rate of wetland loss. In this analysis we use meta-analytic value functions for wetlands and mangroves (Brander et al., 2006; Brander et al., 2012a; Brander et al., 2012b; De Groot et al., 2012) combined with global spatial data on wetland ecosystems to estimate spatially variable (context specific) values. The cost-benefit analysis for this target is then re-calculated to assess the effect of using a more refined value transfer methodology.

This perspective paper focuses primarily on improving estimates of the benefits of avoiding biodiversity and ecosystem loss but there is also a need to address uncertainty in the cost estimates for programmes designed to meet the targets. Footnote 5 in Markandya (2014) notes the issue of high uncertainty and questionable credibility of the cost estimates reported in CBD (2012). These cost estimates require further examination and should be accompanied with some quantification of uncertainty. To this end, we evaluate Target 10 (minimising anthropogenic pressures on coral reefs) to examine the effect of cost uncertainty on the appraisal outcome. The CBD (2012) estimate for the cost of reducing the rate of loss of coral reefs by 50% is USD 648 million for the period 2013-2020. At first sight this seems to be a very small amount given the scale and range of the threats facing coral reefs. Indeed in this case the estimated cost of the envisaged programme may be reasonable but the targeted effect (minimizing anthropogenic pressures on coral reefs) is unlikely to be achieved.

In this perspective paper, we revisit the analysis for Aichi Target 10 and produce new estimates of both the benefits and costs of meeting the target. For the estimation of benefits we use a global database of coral cover (from UNEP World Conservation Monitoring Centre) combined with a meta-analytic value function for coral reef ecosystem services (based on Hussain et al., 2011). For the estimation of costs, we use the same global database of coral cover combined with a meta-analytic value function for costs of Marine Protected Areas (Balmford et al., 2004). The cost-benefit analysis for this target is then re-calculated to assess the effect of using alternative methods for estimating both costs and benefits.

Wetlands

The policy scenario assessed for wetlands is the achievement of Aichi Target 5, which states that "by 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced" (CBD, 2010). The assumptions, parameters and steps used in the assessment of costs and benefits, following those used by Markandya (2014) where relevant, are:

1. The current stock of wetlands, including inland and coastal wetlands, is obtained from the Global Lakes and Wetlands Database (Lehner and Döll, 2004). This includes information on 166,101 wetland ecosystems covering 726 million hectares in 2010.[3] The current stock of mangroves is separately obtained from the UNEP World Conservation Monitoring Centre (described in Giri et al., 2010). This includes information on 5,732 mangrove ecosystems covering 20 million hectares in 2010. The total stock of wetlands included in our analysis is therefore smaller in size than the stock assessed in Markandya (2014), which covered approximately 1,213 million hectares. One reason for this difference is that we do not include lakes and rivers in our definition of wetlands.

2. The baseline annual rate of loss in wetland area is 0.7%, as used in Markandya (2014). This applies to all wetlands (i.e. there is no spatial variation in the rate of loss).

3. The policy scenario annual rate of loss in wetland area is 0.35% (i.e. half of the baseline rate). This applies to all wetlands from 2021-2050 under the policy scenario. The gain in wetland area under the policy scenario vs. the baseline scenario is represented in Figure 1.

4. The time horizon of the analysis is 2013-2050. Initial investment costs are incurred in the period 2013-2020 and benefits (due to the reduced rate of loss) are assumed to accrue in the period 2021-2050.

5. The unit value (US$/ha/year) of ecosystem services provided by each wetland in 2050 is estimated using the meta-analytic value function described in Hussain et al. (2011); and for mangroves the meta-analytic value function described in Brander et al. (2012b) is used. Unit values are estimated separately for the baseline and policy scenarios in order to reflect differences in the scarcity of wetlands under each scenario. The benefits of conservation for each wetland ecosystem is computed as the average of unit values under the baseline and policy scenario, multiplied by the difference in area between the baseline and policy scenario.

6. Lower and upper bound benefits are calculated using the 95% prediction intervals for each wetland site, which are computed using the method proposed by Osborne (2000). The prediction intervals provide an indication of the precision with which the meta-analytic value function can predict out-of-sample values. It is important to note that they do not reflect other sources of uncertainty in the analysis.

7. The global benefits of wetland conservation are computed by aggregating across all wetland ecosystems and over time (for the period 2021-2050) using the assumption

[3] The areal extent of wetlands in 2010 is computed by adjusting the area in 2000 using an annual rate of loss of 0.7%.

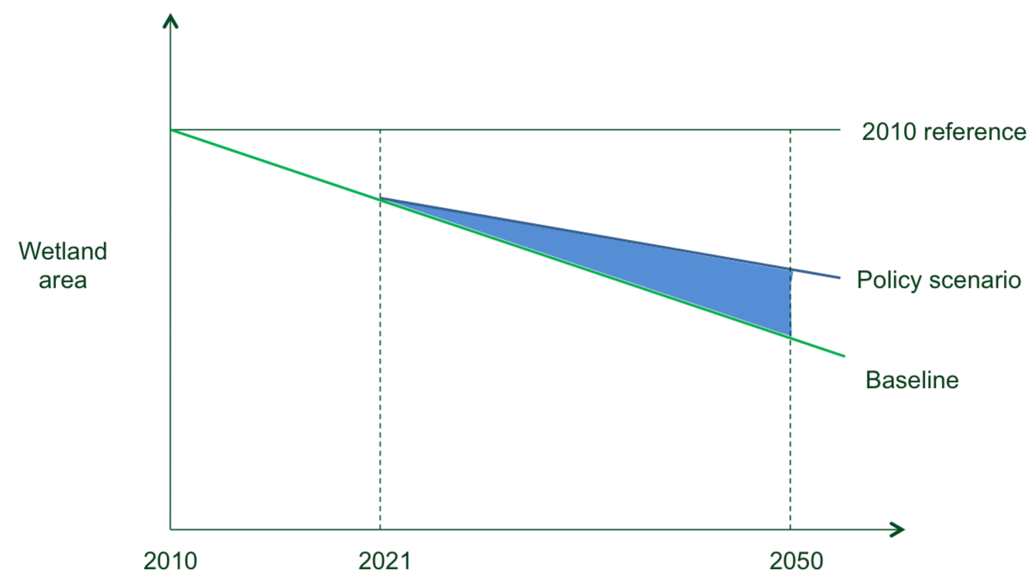
that benefits are distributed linearly over time as the area of wetland conserved increases (at 0.35% per year).

8. Estimates of the lower and upper bound costs of the policy scenario are obtained from CBD (2012). Global lower and upper bound annualised costs for the period 2013-2020 are US$ 39.2 billion and US$ 52.2 billion respectively. We assume that the recurrent costs of the conservation programme continue over the entire time horizon of the analysis. Lower and upper bound annual recurrent costs are US$ 2.87 billion and US$ 3.31 billion respectively.

9. Present value costs and benefits are computed using an annual discount rate of 5%.

The results of the cost-benefit analysis of wetland conservation are presented in Table 1 (corresponding to Table 4 in Markandya, 2014). Following Markandya (2014), four cases are considered: combining the lower bound of the benefit figures and the lower bound of the cost figures; the upper bound of the benefit figures with the lower bound of the costs; the lower bound of the benefits with the upper bound of the costs; and finally the upper bound of the benefits with the upper bound of the costs. In all cases, the conservation programme has a positive benefit-cost ratio (in the range 1.29–2.45) and a corresponding internal rate of return greater than the discount rate used in the analysis (in the range 7–11%). On this evidence, the programme for reducing the rate of wetland loss by half appears to be advisable. In contrast to the cost-benefit analysis results presented in Markandya (2014), the present analysis, using a different approach for transferring values and representing lower and upper bound benefits[4], produces very different results and more clear-cut decision rules.

Figure 1- Gain in wetland area under the policy scenario (represented by shaded area)



[4] In this case the lower and upper bounds refer to the 95% confidence interval and not the minimum and maximum values observed in the literature.

Table 1: Net benefits from Aichi Target 5: 50% reduction in rate of wetland loss

Case	Benefit to Cost Ratio	IRR (%)
Lower Bound of Benefits/ Lower Bound of Costs	1.69	8
Upper Bound of Benefits/ Lower Bound of Costs	2.45	11
Lower Bound of Benefits/ Upper Bound of Costs	1.29	7
Upper Bound of Benefits/ Upper Bound of Costs	1.87	9

Coral Reefs

The policy scenario assessed for coral reefs is the achievement of Aichi Target 10, which states that "By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning." (CBD, 2010). The assumptions, parameters and steps in the assessment of costs and benefits, following those used by Markandya (2014) where relevant, are:

1. The current stock of coral reefs is obtained from the UNEP World Conservation Monitoring Centre. This includes information on 13,682 coral ecosystems covering 21.5 million hectares in 2010. The total stock of coral reefs included in our analysis is therefore slightly larger than the stock assessed in Markandya (2014).

2. The baseline annual rate of loss in coral area is 2.2 %, as used in Markandya (2014). This applies to all coral reefs (i.e. there is no spatial variation in the rate of loss).

3. The policy scenario annual rate of loss in area of coral cover is 1.1 % (i.e. half of the baseline rate). This applies to all coral reefs from 2021-2050 under the policy scenario.

4. The time horizon of the analysis is 2013-2050. Initial investment costs are incurred in the period 2013-2020 and benefits (due to the reduced rate of loss) are assumed to accrue in the period 2021-2050.[5]

5. The unit value (US$/ha/year) of ecosystem services provided by each coral ecosystem in 2050 is estimated using the meta-analytic value function described in Hussain et al. (2011. Unit values are estimated separately for the baseline and policy scenarios in order to reflect differences in the scarcity of coral reefs under each scenario. The benefits of conservation for each coral ecosystem is computed as the average of unit values under the baseline and policy scenario, multiplied by the difference in area between the baseline and policy scenario.

[5] Note that Aichi Target 10 is for anthropogenic pressures to be minimised by 2015. This is unlikely to be achieved and so for the purposes of this analysis we assume that the target will be achieved by 2020 in line with the other Aichi Targets.

6. Lower and upper bound benefits are calculated using the 95% prediction intervals for each coral ecosystem, which are computed using the method proposed by Osborne (2000). The prediction intervals provide an indication of the precision with which the meta-analytic value function can predict out-of-sample values. It is important to note that they do not reflect other sources of uncertainty in the analysis.

7. The global benefits of coral reef conservation are computed by aggregating across all coral ecosystems and over time (for the period 2021-2050) using the assumption that benefits are distributed linearly over time as the area of coral reef conserved increases (at 1.1% per year).

8. Estimates of the lower and upper bound costs of the policy scenario are obtained using a meta-analytic cost function described in Balmford et al. (2004). The cost function relates the annual running costs of marine protected areas (MPAs)[6] to the areal extent of MPAs. We assume that the required areal extent of MPAs to protect coral reefs from anthropogenic pressures is approximately four times the area of coral cover. This assumption is based on a review of coral reef valuation studies and the observation that protected areas necessarily also include other ecosystems (e.g. sea grass, sandy bottom, mangroves etc.) besides areas of coral cover (Brander et al., 2007). Lower and upper bound global annual costs are estimated to be US$ 2.5 billion and US$ 3.5 billion respectively. These cost estimates are at least an order of magnitude higher than those estimated in CBD (2012). We assume that these running costs of MPAs continue over the entire time horizon of the analysis.

9. Present value costs and benefits are computed using an annual discount rate of 5%.

The results of the cost-benefit analysis of coral reef conservation are presented in Table 2. Following the format used for wetlands, four cases are considered: combining the lower bound of the benefit figures and the lower bound of the cost figures; the upper bound of the benefit figures with the lower bound of the costs; the lower bound of the benefits with the upper bound of the costs; and finally the upper bound of the benefits with the upper bound of the costs. In all cases, the conservation programme has a positive benefit-cost ratio (in the range 17.58–33.15) and a corresponding internal rate of return greater than the discount rate used in the analysis (in the range 33 –41%). On this evidence, the programme for reducing the rate of coral loss by half appears to be highly advisable. In this case our results are fairly similar to those of Markandya (2014) and suggest that investing in coral reef conservation yields high returns. Our analysis produced higher estimates of both the costs and the benefits of reducing anthropogenic pressures on coral reefs – but ultimately the result of the cost-benefit is broadly similar.

[6] MPAs enable the protection and restoration of key marine habitats, the replenishment of fish stocks and can enhance the resilience of marine ecosystems. It is recognised, however, that MPAs cannot address all threats facing coral reefs (e.g. rising sea temperature and ocean acidification) and therefore the expansion of MPAs is assumed to result only in a reduced rate of coral loss.

Table 2: Net benefits from Aichi Target 10: 50% reduction in rate of coral loss

Case	Benefit to Cost Ratio	IRR (%)
Lower Bound of Benefits/ Lower Bound of Costs	24.21	37
Upper Bound of Benefits/ Lower Bound of Costs	33.15	41
Lower Bound of Benefits/ Upper Bound of Costs	17.58	33
Upper Bound of Benefits/ Upper Bound of Costs	24.07	37

Discussion and Conclusions

This perspective paper revisits the analysis of conservation targets for wetlands (Aichi Target 5) and coral reefs (Aichi Target 10) by Markandya (2014) using more refined methods for transferring ecosystem services values that account for spatial variation in the determinants of value. The use of meta-analytic value functions (instead of simple unit values) allows ecosystem service values to vary with the area of each ecosystem, the availability of substitutes and the population of beneficiaries living in the vicinity, amongst other variables. Moreover, this approach allows the computation of confidence intervals, which provides a much narrower range of values than the use of extreme minimum and maximum values. Overall this produces a more plausible range of values with which to conduct a cost-benefit analysis and results in more meaningful cost-benefit ratios with which to support decision making. In the case of the wetland analysis, the range of cost-benefit ratios narrows from 0.5–72.7 (see Table 4, Markandya 2014) to 1.29–2.45. In the case of the coral reef analysis, we used meta-analytic value functions to re-estimate both the benefits and costs of achieving a reduction in the rate of coral loss. In this analysis the estimated benefits and costs both increase and the result of the cost-benefit analysis is broadly similar to that of Markandya (2014). We therefore support the conclusion that investing in coral reef conservation is highly worthwhile.

In this perspective paper we have addressed one methodological aspect of conducting global cost-benefit analyses of targets to conserve ecosystem services but there are several others that require further improvement.

In our analysis the rate of ecosystem loss is assumed to be constant both geographically and temporally (e.g. all wetlands decline at a constant annual rate of 0.35% for the period 2021-2050 under the policy scenario), which is unrealistic. Rates of ecosystem loss are likely to be highly variable depending on local anthropogenic pressures that change over time. In subsequent analyses there is a need to examine spatial and temporal variability in rates of loss and the location of conservation efforts since this could have important implications for the costs as well as the benefits of conservation. Such an analysis could make use of available global models of land use change to identify where losses take place.

Producing accurate cost estimates for such broad targets requires in-depth analysis beyond the scope of these challenge or perspective papers. The cost figures presented in CBD (2012) are initial estimates but these should be revisited in greater detail. At the least, this cost information should be critically examined to assess whether it is of the correct order of magnitude (and whether the costed programmes would realistically meet the targets).

Our analysis attempts to improve on the treatment of uncertainty in the value of ecosystem services. As such it only deals with the uncertainty arising from transferring values from primary valuation studies to out-of-sample policy sites.
In addition it is necessary to systematically examine other sources of uncertainty (e.g. measurement errors in primary valuation estimates; biased sampling of available primary valuation studies; consistency and accuracy of spatial data on ecosystem type and extent) recognising that this type of global analysis of costs and benefits of ecosystem change stacks multiple sources of uncertainty.

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