Measuring and Mapping Indices of Biodiversity Conservation Effectiveness

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Abstract

Despite efforts by the 193-member Convention on Biological Diversity and landmark publications on human-environment linkages, global biodiversity loss continues. Climate and biodiversity projections suggest that 40% of all species may face extinction before the end of this century. To enhance issue saliency, many peer-reviewed and grey literature publications have called for interdisciplinary approaches that combine ecological, socioeconomic, and policy measures. Several publications have also recommended that these efforts follow a state-benefits-pressure-response (SBPR) framework.

The present study provides a literature review of effectiveness assessment in biodiversity conservation projects and policies, followed by an example of a composite index that aggregates measures from SBPR categories. Methods of index formulation (weighted linear combination and data envelopment analysis), weighting, and attribute normalization, are demonstrated and compared. Operationalizing the SBPR framework to produce composite indices of linked indicators will integrate interdisciplinary measures in a manner that enhances public understanding and communication of the biodiversity crisis.

Keywords: biodiversity conservation, effectiveness, indices, mapping, interdisciplinary

1 Introduction

The International Year of Biodiversity, 2010, was a politically defining year for global biodiversity conservation. Efforts to significantly reduce the rate of biodiversity loss worldwide by this deadline, a goal set by the Convention on Biological Diversity (CBD, Decision VII/30), were found to have failed (Sachs et al. 2009; Butchart et al. 2010; Larigauderie and Mooney 2010; Mace et al. 2010). Though other global issues such as the continuing economic downturn or climate change may compete for public attention, researchers have suggested that the missed 2010 target is “probably one of the most dramatic policy failures ever, threatening the biological basis of human civilizations” (Spangenberg et al. 2009, p. 9). Indeed, scientists have framed the issue of biodiversity loss as a global crisis (Mooney 2010), citing extraordinary extinction rates (Pimm et al. 1995; Pimm et al. 2006), large-scale loss of species-rich ecosystems (Hoekstra et al. 2005), and potentially dramatic consequences if ecosystem and climactic *thresholds* or *tipping points*, are crossed (Lenton et al. 2008; Ring et al. 2010; SCBD 2010; TEEB 2010c).

The Convention on Biological Diversity (CBD), which entered into force in 1993 and now has 193 parties, was established to address the conservation, sustainable use, and fair and equitable sharing of benefits from biodiversity. To increase social, economic, and political saliency, biodiversity and development initiatives have sought to “mainstream” the “rationale, benefits, and mechanisms for safeguarding ecosystem services” into human behavior through integration with economic sectors that influence land-use (Cowling et al. 2008, p.9486; Swiderska et al. 2008; UNDP and GEF 2008; CBD Decision VI/26). New studies and reports are published regularly on payments for ecosystem services (PES), including carbon credits, which feature prominently in global negotiations over Reduced Emissions from Deforestation and (forest) Degradation (REDD or REDD+) schemes (e.g. Gullison et al. 2007; Miles 2007; Corbera and Brown 2008; Ebeling and Yasué 2008; Combes Motel et al. 2009).

Attention to REDD(+) projects is due in part to the dual role of land-use change as the leading driver of biodiversity loss (Sala et al. 2000; MA 2005; Michelsen 2008; SCBD 2010) and contributor of 12.2% of greenhouse gas emissions (Herzog 2009). Deforestation and other types of land-use change also have important implications for climate change mitigation strategies and biomass carbon storage (MA 2005; IPCC 2007b). Furthermore, urgency in stemming the loss of biodiversity has been stressed due to growing concern over the role of climate change as leading driver of biodiversity loss (Sala et al. 2000; MA 2005; Lee and Jetz 2008; SCBD 2010; Stork 2010). An increase in global mean surface temperature of 1.1–6.4˚C above 1980-1999 levels is likely (> 66% chance) to occur before the end of this century (IPCC 2007a) and new studies suggest that 3.5˚C of warming may be realized as early as 2060 (Betts et al. 2009a & b, adjusted by 0.5˚C following IPCC 2007c). When paired with IPCC (2007c) figures on ecosystem impacts as a function of increasing global average temperature change, these data indicate that today's youngest humans may live to see 40% of all other species go extinct.

Studies critiquing biodiversity assessment suggest that measures of biodiversity must be developed or adapted to be policy-relevant, applicable at multiple scales of biophysical and socio-political organization, and allow for comparison between baseline conditions and targets or goals (Scholes and Biggs 2005; Yue et al. 2007). In addition, several biodiversity initiatives have advocated for, or are working to, adapt indicators to a strategic framework of four general components. These categories of biodiversity attributes are: the condition or state of biodiversity (state), benefits received or derived from biodiversity (benefits), pressures or threats to biodiversity (pressures), and policy responses to manage biodiversity and the impacts upon it (responses) (Mace and Baillie 2007; Maxim et al. 2009; Walpole et al. 2009; McGeoch et al. 2010; UNEP et al. 2010).

One means of integrating all four of these categories into a single measure is through the use of a composite index. Composite indices have been recognized as a useful means to integrate social, political, and economic measurements in development research (Booysen 2002) and are useful in their ability to communicate country-level performance measures to the general public and policy makers (Nardo et al. 2005). They have also been recognized in biodiversity literature as a useful means to communicate broad trends and increase political appeal (Balmford et al. 2005; Buckland et al. 2005; Mace and Baillie 2007).

The most common means of aggregating attributes for environmental and development composite indices is weighted linear combination (also known as weighted average), denoted here as WLC (Booysen 2002; Ratick et al. 2009; and Ratick and Osleeb in review). When applying WLC, importance weights are commonly assigned to attributes in one of two ways: 1) equal weights are applied (computationally equivalent to averaging) due to incomplete knowledge of attributes or their interrelationships; or, 2) subjective weights are applied based on designer preference or 'expert' knowledge. In addition to using arbitrary weights, WLC may conceal the effects of individual, potentially important, attributes once aggregated.

Alternatively, Data Envelopment Analysis (DEA) provides a means of index construction that avoids the use of subjectively-determined weights by employing linear programming to measure a decision making unit's "efficiency" relative to all other decision making units in a cohort (Allen et al. (Allen et al. 1997; Nardo et al. 2005; Ratick et al. 2009). Whereas DEA has previously been used to measure efficiency (Charnes et al. 1978) or vulnerability (Clark et al. 1998), it has been adapted in this study to produce a composite index of measures of effectiveness. Further detail on the indices described in the present study is provided by Wright (2011), on which this paper is based.

The purpose of this research is to operationalize the state-benefits-pressures-responses (hereafter SBPR) framework to enhance issue saliency and interdisciplinary evaluation of biodiversity conservation initiatives. Rather than prescribing yet another uniquely named index comprised of a rigid set of contributing indicators, this exercise is intended to stimulate discussion, learning, and action. To achieve this, the following section (2) will first review recent literature on interdisciplinary assessments of effectiveness and related indices. The following section (3) will then describe example indices that utilize WLC and DEA to incorporate four attributes, each from one of the four SBPR framework categories. In the final section (4), the paper concludes that supporting literature and the demonstrated methodology merit further development of these indices to aggregate linked interdisciplinary attributes into a single communicable measure.

2 Review of conservation effectiveness assessments

2.1 Literature review methods

A search for relevant and recent (i.e. 2000-2010) peer-reviewed literature was conducted using journal databases (Academic Onefile, EconLit, GeoBase, etc.) and Google Scholar with key words such as *global biodiversity, biological diversity, conservation, effectiveness, impacts, 2010 target, CBD, index, indicator, social, cultural, political, values, ecosystem services, REDD, land-use,* and *climate change*. Recent publications related to biodiversity policy from leading international governmental and non-governmental organizations, as well as procedural notes on CBD decisions and meetings, were also reviewed. Reference literature cited by sources identified in this search were also reviewed and included if relevant.

Of the more than 300 sources reviewed, 223 were selected for inclusion in this study based on their relevance to effectiveness assessment, index use or construction, and to provide background or supporting information (see Wright 2011). Of the selected publications, 75 were found to be directly relevant to the review of interdisciplinary assessment of biodiversity conservation effectiveness, and 27 were found to provide examples of composite indices or insight into their use. Three publications were included in both categories. In the interest of clarity, definitions of the key terms: *effectiveness*, *indicator*, *composite environmental indicator*, *attribute*, *environmental index*, and *composite index* are provided in Wright (2011) and applied herein.

2.2 Assessments of protected areas, social, economic, and policy effectiveness

Selected peer-reviewed studies with direct relevance to interdisciplinary assessment of conservation effectiveness or impact are presented in Table 1, grouped by effectiveness type. Four distinct types of effectiveness assessments were identified within the literature: protected areas, economic, social/cultural, and policy/institutional. For each of the four effectiveness types identified in Table 1, four example studies are listed with brief descriptions of their assessment perspective (ex ante or ex post), scale, methods, and findings. The 16 studies included therein are intended to be representative rather than exhaustive and are further described below (see Wright 2011 for greater detail). Literature on interdisciplinary assessment using indices is reviewed subsequently.

Table 1. Representative studies of effectiveness in biodiversity conservation initiatives.

	Recent Examples	Ex Ante/ Ex Post	Evaluation Methods/ Data	Main finding(s) or conclusion(s)
Protected Areas Effectiveness	Bruner et al. 2001	Ex post	Survey of PA managers and indicator correlations	PAs are effective and should remain a central conservation strategy
	Rodrigues et al. 2004	Ex post	PA distribution models and gap analysis	PA expansion needs to better incorporate biodiversity patterns at genetic, species, and ecosystem levels
	Oliveira et al. 2007	Ex post	Remote sensing of land cover change inside vs. outside PA	Moderate success in protecting forests can be sustained through balanced use, protection, and law enforcement
	Andam et al. 2008	Ex post	Remote sensing of land cover change inside vs. outside PA and other GIS data	Conventional evaluation of PAs overestimates their effectiveness due to biases (see text)

Continued on the following page.

Table 1. Continued

	Recent Examples	Ex Ante/ Ex Post	Evaluation Methods/ Data	Main finding(s) or conclusion(s)
Economic Effectiveness	Nelson et al. 2009 and others	Ex ante	Econometric model of land use/ change; models of ecosystem services, biodiversity, and commodity production; efficiency frontier	Mixed results on policy impacts on ecosystem services and performance of targeted vs. general policies; components of InVEST are new and untested
	Crossman and Bryan 2009	Ex ante	Spatial optimization of restoration cost-effectiveness	Further demonstrates that spatial optimization models can improve environmental benefits and minimize costs
	Laycock et al. 2009	Ex post	Survey of project managers and cost-effectiveness analysis (CEA) of targets	Demonstrates CEA in conservation and finds no cost-effectiveness correlation
	Wikberg et al. 2009	Ex post	Unweighted average of nine ranked, cost-effectiveness parameters	Field surveys were more effective than satellite methods of conservation area selection
Social/ Cultural Effectiveness	Corbera et al. 2007	Ex post	Interviews, surveys of stakeholders, and calculation of payments	PES projects can effectively reduce PA costs and increase participation, particularly community-based initiatives
	Kessler et al. 2007	Ex post	Categorizes impact patterns between biodiversity and socioeconomic indicators (e.g. both declining)	Biodiversity and socioeconomic decline found in 54% of assessed production areas; challenges export development
	Steffan-Dewenter et al. 2007	Ex post	Compares ecological indicators and socioeconomic drivers derived from field surveys, censuses, and interviews	Finds cultural influence on land use and low-shade agroforestry as the best economic-ecological compromise
	Waylen et al. 2010	Ex post	Systematic review of community-based conservation projects using socioeconomic and ecological indicators	Local context and engagement influenced outcomes but participation had limited impact
Policy/ Institutional Effectiveness	Mas and Dietsch 2004	Ex post	Comparison of certification criteria with biodiversity management indicators	Shade coffee certification programs identified farms of conservation significance
	Donald et al. 2007	Ex post	Comparison of species population trends pre- and post-policy implementation	Rare/vulnerable bird populations increased after implementation of the EU Birds Directive with a 10-year lag
	Duit et al. 2009	Ex post	Tests for correlation between indicators of biodiversity, social capital, governance, and policy	Social capital correlated negatively with diversity. Policy correlated positively with density but not diversity.
	Cóndor et al. 2010	Ex ante	Applied a Multicriteria Decision Aid model to assess potential project performance	Project are categorized by level of international convention synergy; agroforestry was synergistic

Protected Areas Effectiveness

In general, the literature critiquing the effectiveness of conservation strategies based on protected area (PA) establishment and management addresses three broad subject areas: assessment of outcomes, improvements to assessment methodology, and alternatives to PA approaches. The quantitative studies included in Table 1 that assess outcomes of protected area approaches suggest that benefits of these areas have included reduction of anthropogenic disturbance and deforestation (Andam et al. 2008), and mitigation of other threats such as logging and hunting (Bruner et al. 2001). For example, satellite-based assessment of forest disturbance in the Peruvian Amazon by Oliveira et al. (2007) indicated that only 1% of deforestation and 2% of disturbance occurred within protected areas. Another supporting assessment of Costa Rican PA effectiveness in terms of avoided deforestation found that 11% of PAs protected by 1979 would have been deforested by 1997 if not protected (Andam et al. 2008).

However, these documented benefits come with caveats. In terms of species conservation, at least 12% of species included in the IUCN Global Mammal Assessment are not represented in any protected area (Rodrigues et al. 2004). Andam et al. (2008) also point out that conventional assessments (including Oliveira et al. 2007 and Bruner et al. 2001) of avoided deforestation/disturbance overestimate the impact of PAs by a factor of 3 or more. Other case studies also suggest that devolution of authority to communities may be an effective alternative to the top-down establishment and management of PAs (Oliveira et al. 2007; Xu and Melick 2007; Behera 2009).

Economic Effectiveness

Within the last ten years, an abundance of literature has been published on an alternative to the classic protected areas model, payments for ecosystem services (e.g. Murtough et al. 2002; UNEP 2004; Casey and Boody 2007; Ranganathan et al. 2008; Voora and Venema 2008; SCBD 2009). The economic studies presented in Table 1 provide examples of how cost-effectiveness assessments can be incorporated in biodiversity conservation. Spatially-explicit economic studies in this table employ systematic GIS methodologies to map cost-effective restoration sites (Crossman and Bryan 2009), and to model policy effects on ecosystem services, commodities, and biodiversity (Nelson et al. 2008; Nelson et al. 2009; Tallis and Polasky 2009). Crossman and Bryan (2009) demonstrate their model's potential to provide air, GHG, and biodiversity benefits and minimize economic impacts to agriculture in Australia. Demonstrating another spatially-explicit method, Nelson et al. (2009) and others employ an econometric model (InVEST) to project ecosystem service provision, commodity production, and biodiversity conservation under three different land use scenarios.

The other two economic studies presented in Table 1 provide examples of policy cost-effectiveness evaluation without an explicitly spatial component. Laycock et al. (2009) evaluate the UK Biodiversity Action Plan using cost-effectiveness assessment (CEA), which is described as less data-intensive than cost-benefit analysis. Though not identified as an index, the non-spatial economic assessment by Wickberg et al. (2009) is derived from an unweighted average of nine attributes (essentially WLC with equal weights).

Social-Cultural Effectiveness

The importance of incorporating local knowledge and values to biodiversity conservation is increasingly being recognized (Pretty and Smith 2004) and the socio-cultural studies presented in Table 1 provide several examples of impact studies that incorporate related attributes. Two of the studies listed in this table examine the role of economic and cultural factors on land use in agro-commodity production (Steffan-Dewenter et al. 2007) and challenge the socio-ecological benefits of export-oriented development (Kessler et al. 2007). Integrating several methods for deriving ecological and socioeconomic indicators, Steffan-Dewenter et al. (2007) compare impacts of agroforestry land use strategies and document the influence of an immigrant ethnic group on land cover change. Kessler et al. (2007) identify negative impact patterns of agro-commodity development on a combination of six socioeconomic indicators and the Natural Capital Index (derived from GLOBIO3 of Alkemade et al. 2009), for four developing countries (Argentina, Brazil, Honduras, and Vietnam).

The other studies listed in the Table 1 further emphasize the need for understanding of local context and engagement. Focusing on equity implications of ecosystem services within protected areas versus rural communities, Corbera et al. (2007) find that rural initiatives are better able to integrate service providers in management decisions and that marketing ecosystem services in protected areas excludes resource users from development benefits. Waylen et al. (2010) provide a broader assessment of 15 indicators extracted from 68 community-based conservation studies using a coding protocol.

Policy Effectiveness

Although there are many legal and institutional sources of biodiversity-related regulations ranging from treaties, statutes, and case law, to customary practices and religious norms (Glowka et al. 1998), challenges remain in assessing their effectiveness. For example, the 2004 biodiversity programs study of the Global Environment Facility reports that less than 20 of the 141 projects reviewed reported impacts of any kind and fewer still provide data that can be used to derive trends (Dublin et al. 2004). Some studies of policy/institutional effectiveness have been conducted, however, and examples are provided in Table 1.

Through the use of biodiversity indicators at local farms in Mexico, Mas and Dietsch (2004) verify the effectiveness of coffee certification programs in preserving biodiversity. Working across scales, Cóndor et al. (2010) provide a multicriteria decision aid to assess synergies at the project level between the three international Rio conventions (the CBD, the UNFCCC, and the UNCCD) for ten afforestation/reforestation projects developed under the Kyoto Protocol's Clean Development Mechanism. Assessing policy effectiveness across Europe, Donald et al. (2007) compare population data for rare and vulnerable birds from before and after implementation of the EU's Bird Directive. Duit et al. (2009) also assess bird species as an indicator of biodiversity, and test for correlation with measures of social capital, governance, and policy performance in 20 European countries.

Summary of Findings on PAs, Economic, Social, and Policy Effectiveness

Many of the examples provided in Table 1 make clear efforts to incorporate interdisciplinary analysis. Assessments of PA effectiveness suggest that while there are many opportunities to improve upon management and community involvement, PAs remain a central component of biodiversity conservation. Related interdisciplinary research emphasizes that socioeconomic factors should be included in conservation, including PA design and administration. Ecosystem services and modified forms of economic evaluation have proven to be popular vehicles for bridging ecological and socioeconomic issues but more time will be required to evaluate newly developed approaches and their impacts.

Studies focusing on socio-cultural aspects of conservation projects stress the importance of local context, knowledge, and engagement but provide mixed assessment of the success of integrative projects. Similarly, policy assessments reviewed herein demonstrate that while there are specific examples of policy effectiveness, links between policy, social capital, and conservation outcomes remain unclear. Lack of balance and connectivity in assessment perspective (ex post or ex ante) is also evident in the examples provided. Of the 16 studies included in Table 1, 13 provide assessments with the intent to identify past or current projects, policies, or components thereof as effective or ineffective in achieving desirable outcomes (ex post). In contrast, only three of the examples provide assessments with the intent to directly inform or facilitate effective projects in the future by identifying requisite conditions or criteria (ex ante).

2.3 Biodiversity indices used in effectiveness assessments

Focusing specifically on composite indices, Table 2 provides a summary of all such examples identified in the literature review. For the purposes of this study, composite indices are defined as macro-level indices that measure multi-dimensional concepts by aggregating interdisciplinary (e.g. environmental, social, economic, political) attributes (Booysen 2002; Nardo et al. 2005). These measures are listed in order of the number of composite variables employed when aggregating the composite index and in sequential/chronological order within one group of similar indices (Biodiversity Intactness Index). For each index in the table, information is also provided on the SBPR categories to which their attributes correspond, assessment perspective, method used to select attributes and weights, method of aggregating index values, and the index source publication(s). Further index description is provided here, with greater detail provided by Wright (2011).

Table 2. Indicators and indices used in biodiversity conservation.

Indicator or Index Name	Num. Var.[a]	Category S	B	P	R	Ex Ante/ Ex Post	Selection Method[b]	Weighting Method	Aggregate Format	Source(s)
GLOBIO3	1 (1)	x		x		Either	Ad hoc	Area	Sum	Alkemade et al. 2009
Index of Biocultural Diversity (IBCD)	2 (2)	x	x			Ex ante	Ad hoc	Equal/ average	Sum	Loh and Harmon 2005
Biodiversity Intactness Index (BII)	3 (1)	x		x		Ex post	Ad hoc	Species rich & area	Quotient of sums (%)	Scholes and Biggs 2005; Rouget et al. 2006
Biodiversity Intactness Variance (BIV)	4 (1)	x		x		Ex post	Ad hoc	Species rich & area	Quotient of sums (%)	Hui et al. 2008
SARII (see text)	2 (1)	x		x		Ex post	Ad hoc	Species richness	Sum	Faith et al. 2008
GACII (see text)	2 (1)	x		x		Ex post	Ad hoc	Species richness	Sum	
Red List Indices (RLIs)	3 (1)	x		x		Ex ante	Ad hoc	Threat category (0-5)	1 - quotient of sum over max value	Butchart et al. 2005; Butchart et al. 2007
National Biodiversity Risk Assessment Index (NABRAI)	3 (3)	x		x	x	Ex ante	Ad hoc	Equal/ average	Quotient	Reyers et al. 1998; Reyers and James 1999
Biodiversity efficiency	4 (1)	x		x	x	Ex post	Ad hoc	DEA	Sum	Halkos and Tzeremes 2010
GEF benefits index for biodiversity (GBI_{BIO})[c]	4 (2)	x		x		Ex ante	Ad hoc	Arbitrary	Sum	GEF 2005; Dev Pandey et al. 2006; World Bank 2008b
Natural Resource Management Index (NRMI)[cd]	4 (2)		x		x	Ex ante	Ad hoc	Arbitrary	Sum	CIESIN 2009
Priority conservation areas	4 (4)	x		x	x	Ex ante	Ad hoc	Equal/ average; arbitrary	Sum	Shi et al. 2005
Return on Investment (ROI)	7 (1)	x		x	x	Ex ante	Ad hoc	Equal/ average; arbitrary	Sum	O'Connor et al. 2003
Environmental Performance Index (EPI)[cd]	10 (10)	x	x	x	x	Ex post	Ad hoc	Arbitrary	Sum	Emerson et al. 2010
Austrian Forest Biodiversity Index (AFBI)	13	x			x	Either	Expert advisory board	Expert advisory board	Sum	Geburek et al. 2010

Framework categories refer to state (S), benefits (B), pressures (P), and responses (R). Table design is based on Booysen (2002). Notes: [a] The number of variables that are composite measures themselves are shown in parentheses; [b] Ad hoc refers to expert opinion and/or intuitive appeal as per Booysen (2002); [c] No supporting peer-reviewed publications were found; [d] The NRMI and EPI were not designed as biodiversity-focused indices.

Although the first index listed in Table 2 is described as aggregating one weighted attribute (Alkemade et al. 2009), the attribute is a composite of five raster images of mean species abundance that are each derived using cause-effect relationships for drivers impacting biodiversity. The second composite index listed, created by Loh and Harmon (2005), provides an index of biocultural diversity (IBCD) to measure the global variety of natural and cultural systems. Comparative analysis between the IBCD and the index example of this study is detailed in Wright (2011).

The subsequent group of indices all relate to the Biodiversity Intactness Index (BII). The original BII (Scholes and Biggs 2005) is described as a species richness- and area-weighted index of land use impacts on species groups (Rouget et al. 2006). A related measure of Biodiversity Intactness Variance (BIV) was developed to support the BII by measuring its uncertainty (Hui et al. 2008) but adds no SBPR attributes to its calculation. Building on the BII, the Species-Area Relationship Intactness Index (SARII) and Genetic-diversity Abundance-fraction Curve Intactness Index (GACII) were developed to reflect variation in species and genetic diversity, respectfully (Faith et al. 2008).

Another species-focused index, the Red List Index (RLI), incorporates both the number of species assessed and their threat status to track changes and project future extinction risk for sets of species (Butchart et al. 2005; Butchart et al. 2007). In its improved form, the RLI differs from the other indices in that it is a proportion or quotient that has been adjusted to reverse its scale such that lower proportions result in higher RLI values (RLI = 1 – quotient).

The next two indices on the table both incorporate state, pressure, and response attributes. The National Biodiversity Risk Assessment Index (NABRAI) was designed to identify countries of “critical conservation concern” (Reyers et al. 1998; Reyers and James 1999). Taking a unique approach, Halkos and Tzeremes (2010) provide the first application of biodiversity-related DEA, to measure the efficiency of countries’ biodiversity policies and the impacts of four external variables on them.

Found within the grey literature, the GEF benefits index for biodiversity (GBI_{BIO}) and the Natural Resources Management Index (NRMI), each incorporate two components of the SBPR framework. Providing a measure of “biodiversity potential” at the country-level, the GBI_{BIO} combines weighted sub-indices of species and ecosystem representation and threat (GEF 2005; Dev Pandey et al. 2006; World Bank 2008b). Applying a different approach, the NRMI combines proximity-to-target attributes for four environmental attributes, the results of which are used to support selection of poverty reduction projects by the Millennium Challenge Corporation (CIESIN 2009).

Two studies listed in Table 2 are similar both in their incorporation of attribute types and in referencing Mittermeier et al.’s (1997) megadiversity countries priority template. Shi et al.’s (2005) index of priority conservation areas combines several social and ecological indicators to prioritize future conservation action among the megadiversity countries and biodiversity hotspots (Mittermeier et al. 1999; Myers et al. 2000). Meanwhile, the Return on Investment index (ROI), combines attributes in an indicator of economic cost to inform decision making and finds limited overlap between the resulting index ranks and the megadiversity countries template (O'Connor et al. 2003).

One of the largest indices (in number of variables) in Table 2, the Environmental Performance Index (EPI), is the only composite index found that incorporates all four attributes of the SBPR framework. However, the EPI is designed to rank countries on their performance in terms of both environmental public health and ecosystem vitality, with biodiversity contributing only 4.2% of final index values (Emerson et al. 2010). The final index on Table 2, the Austrian Forest Biodiversity Index (AFBI), combines nine state and four response attributes (Geburek et al. 2010) and includes genetic parameters.

Summary of Findings on Composite Biodiversity Indices

Of the 13 distinct composite indices identified here, nearly all (12) included at least one variable that was also a composite measure and nearly all (12) were comprised of attributes that were selected ad hoc. Five composite indices were found to incorporate at least three of the SBPR categories and only one composite index (the EPI) was found to incorporate all four components. Three composite indices were found to include benefits attributes in their calculation (IBCD, NRMI, and EPI), roughly half included response attributes, and a majority (10) incorporated threats attributes. Roughly half of the composite indices (seven) were applied toward ex ante analysis (priority setting), four were applied toward ex post analysis, and two were designed to be applied toward either perspective. Neither of the latter two indices, however, was used to compare goals with realized outcomes. Geburek et al. (2010) provide the only example of attribute selection explicitly designed to facilitate both target setting ex ante and outcomes assessment ex post.

For all 13 composite indices, selection of index attributes was performed by experts, and in nearly all cases (12) this was done ad hoc by those designing the index. The AFBI was the notable exception in that attributes and weights were selected by an expert advisory board. Little consensus was found in the specific method used to apply weights ad hoc. The most common methods of applying weights were arbitrary assignment of unequal weights (5 indices) and equal/average weights (4 indices). Lastly, the most common aggregation method was summation (10 indices) followed by methods that employed a quotient (3 indices).

3 A SBPR index of biodiversity conservation effectiveness

3.1 Index data

The SBPR index examples described herein were designed to identify (ex ante) administrative units that have the highest (and lowest) composite scores based on recent measures of key attributes related to climate-biodiversity initiatives targeting carbon storage (such as REDD or REDD+). No global data sources were identified in this study that would allow for performance assessment (ex post) of attributes from all four SBPR categories at two consistent points in time. Index calculation was conducted at the global scale with a terrestrial focus to accommodate comparison between politically-relevant administrative units and with related studies, and to align with the scale and ecosystems in which benefits of biomass carbon storage operate. A summary of data parameters is presented in Table 3, see Wright (2011) for further detail.

Table 3. Summary of global source data.

	State	**Benefits**	**Pressures**	**Responses**	**Geo. Boundaries**
Data	Vertibrate endemism richness (ER)	Biomass carbon storage (CS)	Land cover change (LCC)	Terrestrial areas protected (PA)	Administrative boundaries
Type	Vector shapefile	Raster image	Raster image	Spreadsheet	Vector shapefile
Resolution	90 ecoregions	~1 km^2	25 km^2	219 admin. units	247 admin. units
Units	Range equivalents per 10,000 km^2	0.01 metric tons per ha	17 land cover classes	km^2	km^2
Source	Kier et al. (2009)	Ruesch and Gibbs (2008)	MODIS Land Cover (MCD12Q1)**	MDG indicator website*	Global Admin. Areas website***
Date	2009	2000	2001 and 2007	2009	28-Apr-2010

*Dates listed indicate the year published, recorded, or when data were obtained. Notes: *http://unstats.un.org/unsd/mdg/; **http://webmap.ornl.gov/wcsdown/; *** www.gadm.org*

To obtain attribute values at an international and politically-relevant scale, data were summarized at the level of administrative units (typically countries or territories) obtained from the GADM database of global administrative areas. This dataset provides highly detailed boundaries for 247 global administrative units (GAUs). Data selected to represent each of the four linked indicator categories (state, benefits, pressures, and responses) are briefly described here. For detailed description, see Wright (2011).

State – Endemism richness (ER): Kier et al. (2009) provide a measure of endemism richness representing the total number of species range equivalents contained within each of 90 ecoregions. This measure was selected to represent biodiversity as it incorporates both richness and endemism, measures the conservation value of an area, and is sensitive to the effects of invasive species (Kier et al. 2009). The Kier et al. (2009) ER data were recalculated at the administrative unit level through shapefile intersection in ArcMap 9.3.1 (ESRI).

Benefit – Carbon storage (CS): Biomass carbon storage value was chosen to represent the benefit attribute in this study following the biodiversity-carbon linkage described earlier. Global raster data on above- and below-ground biomass carbon storage was obtained from Ruesch and Gibbs (2008). Individual cell (pixel) values for the raster image of global biomass carbon were summed for each administrative unit using zonal statistics in ArcMap 9.3.1.

Pressure – Land cover change (LCC): Changes in land cover were quantified for the two most human-intensive land cover categories identified in the International Geosphere-Biosphere Programme Data and Information System (IGBP-DIS). These categories were "urban and built-up" and "croplands." Land cover change between the 2001 and 2007 raster images was quantified using Idrisi Land Change Modeler (Clark Labs) and the sum of gained cells per GAU was then calculated through zonal statistics in ArcMap 9.3.1.

Response – Protected area (PA): Protected area coverage is used in this study as a proxy indicator of political will for conservation action and was also selected due to its use in both the Millennium Development Goals (MDGs) and Biodiversity Indicator Partnership (BIP) indicators. Though protected areas are not eligible for carbon credits for reducing deforestation, other incentives such as credits for sustainable management may be applicable (World Bank 2008a).

3.2 Index formulation and evaluation methods

As described above, index formulation was designed to measure potential or ex ante conservation effectiveness, which is assumed to be positively associated with each of the four attributes. That is, the potential for effective conservation action is assumed to be high in areas which have the greatest quantity of biodiversity, the greatest quantity of benefit value, the greatest threat from pressures, and where policy responsiveness is high. To prepare data for index calculations, ER, CS, and LCC data were processed to obtain values for each of the four attributes within each administrative unit. Values for small outlying territories were consolidated under sovereign nations where necessary to obtain consistency with the PA data. Administrative units were excluded from the analysis where sovereignty was disputed, one or more datasets were absent, or if their surface area was smaller than the 25 km^2 resolution of the LCC data.

Once values were obtained for all four attributes (ER, CS, PA, and LCC), they were normalized through log transformation. Following Loh and Harmon (2005) and Harmon and Loh (2004), two methods of normalizing data were applied and compared, richness-normalization and area-normalization. To normalize by richness, the log of each attribute value for an administrative unit was divided by the log of the global total for that attribute. To normalize by area, nonlinear regressions of log attribute versus log area were performed to establish deviation from expected values based on the concept of the species-area relationship (modified from Groombridge and Jenkins 2002; Harmon and Loh 2004; Loh and Harmon 2005). For calculation details and equations, see Wright (2011).

Normalized attribute values were then scaled 0-1 to input into WLC and DEA indices. The mathematical formulation of WLC and DEA followed Ratick et al. (2009) and Ratick and Osleeb (in review). This method assumes that the DEA output values (denominator) are equal to 1 and uses DEA to construct scalar index values of a multidimensional concept. To produce the WLC index, all four attributes were given equal weight (0.25) and summed. WLC output values were also scaled to a maximum value of 1 to enable correlation comparison with the DEA output, which ranges 0-1 by design.

DEA was conducted three times for each normalization method, using three different weight constraints, or assurance regions. The three DEA assurance regions chosen were: no assurance region (weights are constrained to be ≥0%), ≥10%, and ≥25%; hereafter denoted simply as DEA_0, DEA_{10}, and DEA_{25}. These assurance regions were selected to facilitate comparison between the WLC results and an unconstrained DEA (0%), a DEA weighted somewhat similarly to the WLC (25%), and a 'compromise' DEA (10%) that ensured all attributes contributed to the final index.

To produce global cloropleth maps (Mollweide projection), index values were joined to the GAU datalayer (Figure 4.6). Finally, a subset of the top 20 richness- and area-normalized index scores from the WLC and DEA formulations were selected for further examination in accordance with Ratick and Osleeb (in review) and Loh and Harmon (2005).

3.3 Index results

Applying the methods of index formulation described above and by Wright (2011), attribute and index values were obtained for a total of 216 global administrative units. Final index values for all eight formulation methods are illustrated by normalization method in Figures 1 and 2 as scatter plots. These figures illustrate the tendency of DEA to produce higher index scores than WLC, which is expected given DEA's optimization approach. The minimum percent increase of DEA above WLC was 0.0% for both normalization types, indicating that DEA values were always equal to or greater than WLC values for the same administrative unit.

Final index WLC-Area and DEA_{10}-Area values are mapped in Figures 3 and 4, respectively. Color categories were assigned by decile to compensate for differences in the frequency distributions of indices and attributes (see Wright 2011) and to facilitate comparison between maps. Similar maps of richness-normalized values are not included due to high correlation with administrative unit size (see Wright 2011), which can already be compared visually in these maps.

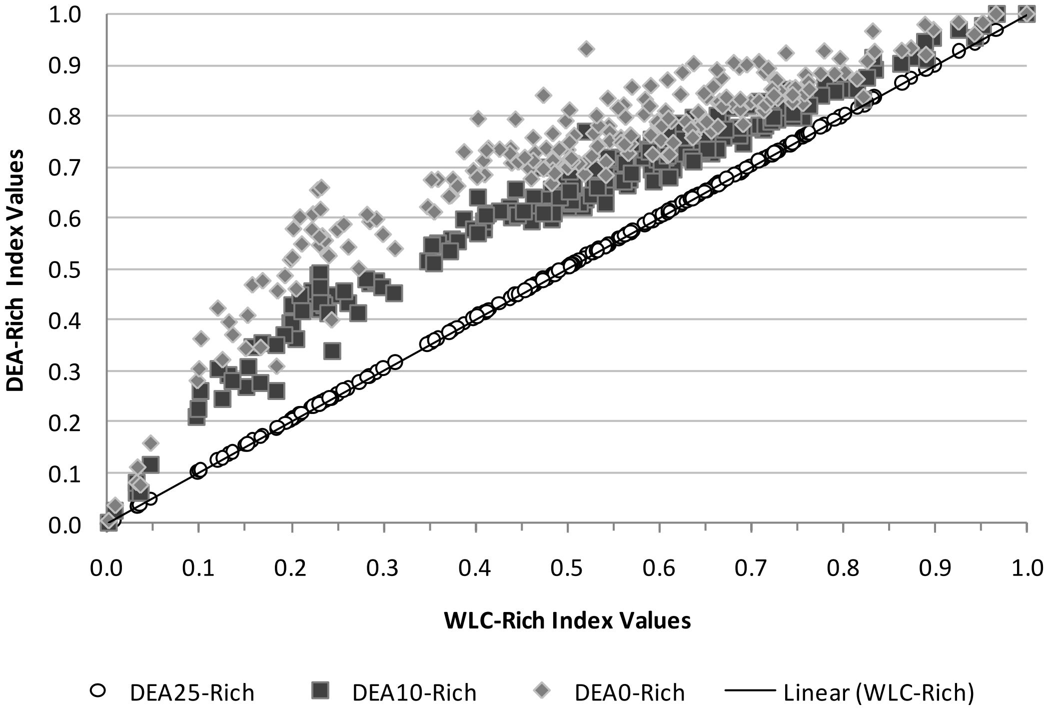

Figure 1. DEA-Rich versus WLC-Rich index values.

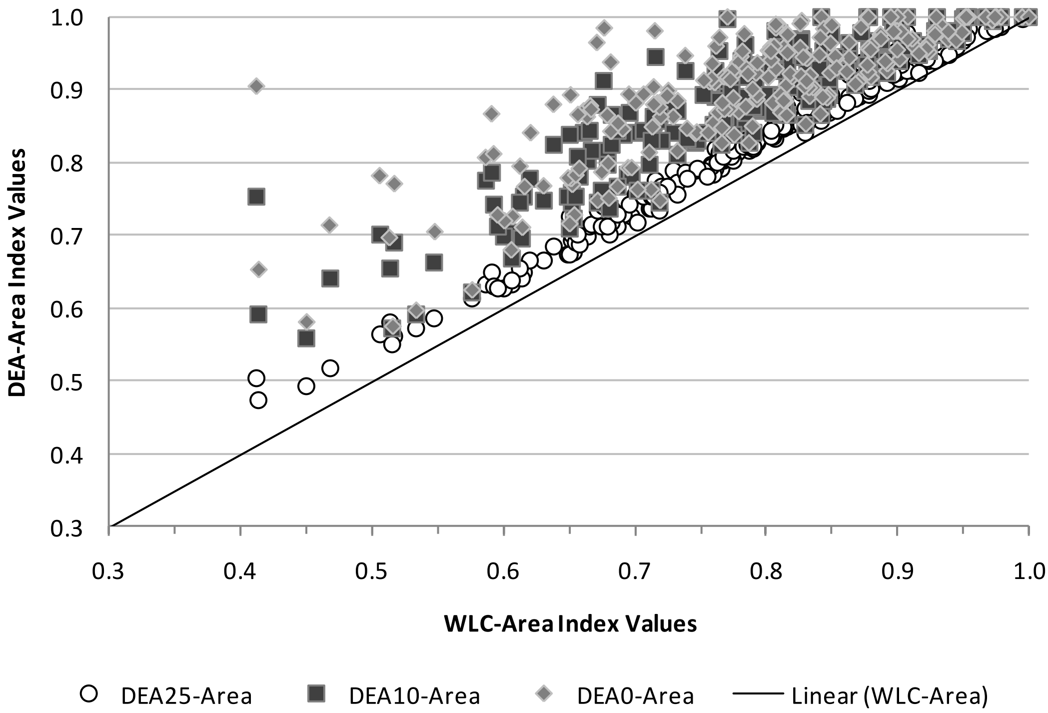

Figure 2. DEA-Area versus WLC-Area index values.

Shifting the index evaluation to a more detailed analysis of the top 20 richness- and area-normalized WLC scores, Table 4 lists the top 20 administrative units by RLC-Rich values along with their corresponding DEA10-Rich values, size, and identifies megadiversity countries (from Mittermeier et al. 1997). Compositionally, the top 20 administrative units ranked by WLC-Rich and DEA_{10}-Rich values are nearly identical. 11 of the 17 megadiversity countries are found within the top 20 WLC-Rich ranks and all of these top 20 ranks are found within the largest 33 countries. 12 megadiversity countries were found within the top 20 DEA_{10}-Rich ranks.

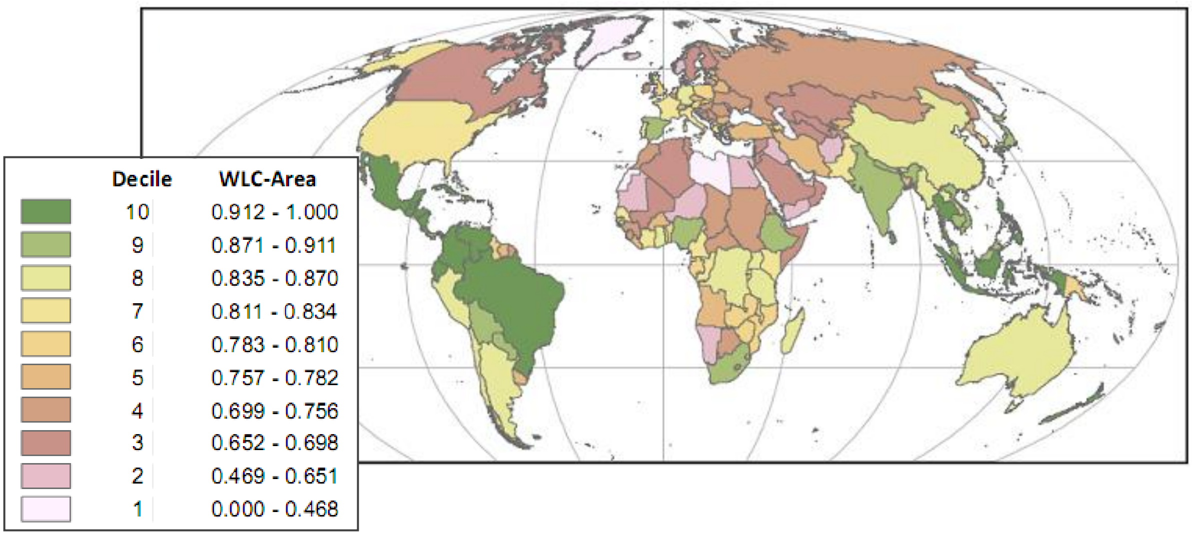

Figure 3. Global map of final index values by decile for WLC-Area.

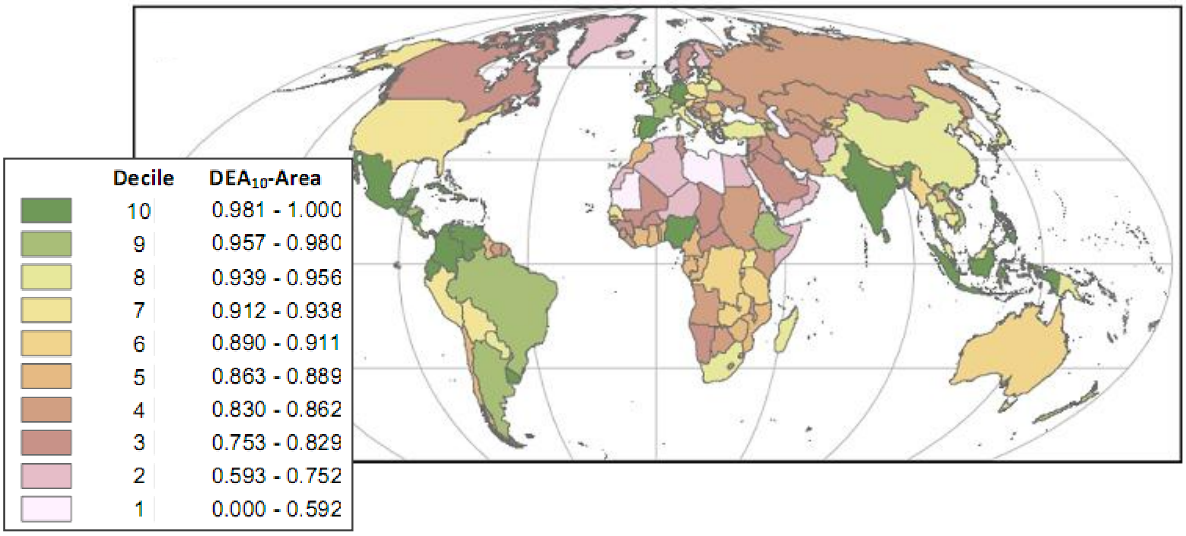

Figure 4. Global map of final index values by decile for DEA_{10}-Area.

Table 4. Top 20 WLC-Rich index scores with corresponding DEA10-Rich scores and administrative unit size.

Global Administrative Unit	WLC-Rich (rank)	DEA_{10}-Rich (rank)	Size (rank)
Brazil	*1.000 (1)*	*1.000 (2)*	*0.958 (5)*
China	*0.967 (2)*	*1.000 (1)*	*0.965 (4)*
United States	*0.952 (3)*	*0.977 (3)*	*0.966 (3)*
Australia and territories	*0.944 (4)*	*0.953 (6)*	*0.952 (6)*
Russia	0.926 (5)	0.968 (4)	1.000 (1)
India	*0.899 (6)*	*0.954 (5)*	*0.901 (7)*
Mexico	*0.891 (7)*	*0.909 (10)*	*0.870 (15)*
Indonesia	*0.890 (8)*	*0.944 (7)*	*0.867 (16)*
Argentina	0.874 (9)	0.915 (8)	0.891 (8)
Canada	0.865 (10)	0.903 (11)	0.968 (2)
Colombia	*0.835 (11)*	*0.890 (12)*	0.837 (26)
Dem. Rep. of the Congo	*0.833 (12)*	*0.914 (9)*	*0.881 (12)*
Venezuela	*0.827 (13)*	*0.875 (13)*	0.824 (33)
Ethiopia	0.822 (14)	0.831 (22)	0.835 (27)
South Africa	*0.815 (15)*	*0.850 (18)*	0.842 (25)
Nigeria	0.803 (16)	0.853 (16)	0.825 (32)
Bolivia	0.798 (17)	0.867 (15)	0.835 (28)
Sudan	0.791 (18)	0.847 (19)	0.885 (10)
Peru	*0.780 (19)*	*0.869 (14)*	*0.845 (20)*
Tanzania	0.776 (20)	0.841 (20)	0.826 (31)

Size is calculated as log area, scaled to a maximum value of 1. Rank values are shown in parentheses. Megadiversity countries (Mittermeier et al. 1997) within top 20 ranks of each column are shown in italics.

Table 5 is similar to the previous table but shows the top 20 administrative units based on WLC-Area. In contrast to the previous table, 13 of the top 20 DEA_{10}-Area ranks overlap with the top 20 WLC-Area ranks. Size ranks shown on this table cover a broad range of 15-210. 6 of the 17 megadiversity countries are found among the top 20 WLC-Area ranks, and 6 are also found among the top 20 DEA_{10}-Area ranks (not all shown here). Summarizing overlap between the top 20 ranks of WLC-Rich, DEA_{10}-Rich, WLC-Area, and DEA_{10}-Area, only three administrative units were found in all four index methodologies: Venezuela, Indonesia, and Colombia. Fewer overlaps were found between top 20 ranks of the same formulation method (4 for WLC, 3 for DEA_{10}) than between top 20 ranks of the same normalization type (19 for richness, 13 for area).

Table 5. Top 20 WLC-Area index scores with corresponding DEA10-Area scores and administrative unit size.

Global Administrative Unit	WLC-Area (rank)	DEA_{10}-Area (rank)	Size (rank)
Sri Lanka	1.000 (1)	1.000 (8)	0.666 (122)
Dominican Republic	0.995 (2)	1.000 (14)	0.648 (130)
Ecuador	*0.979 (3)*	*1.000 (9)*	0.748 (77)
Cuba	0.975 (4)	1.000 (7)	0.697 (105)
Philippines	*0.974 (5)*	*1.000 (1)*	0.757 (72)
Guatemala	0.972 (6)	0.994 (19)	0.696 (106)
Nicaragua	0.967 (7)	1.000 (15)	0.707 (97)
Trinidad and Tobago	0.954 (8)	1.000 (10)	0.513 (169)
Venezuela	*0.952 (9)*	*1.000 (3)*	0.824 (33)
Martinique	0.952 (10)	1.000 (13)	0.422 (178)
Indonesia	*0.950 (11)*	*0.997 (17)*	*0.867 (16)*
Jamaica	0.949 (12)	0.979 (26)	0.559 (164)
Honduras	0.945 (13)	0.968 (34)	0.698 (102)
Mexico	*0.939 (14)*	0.980 (22)	*0.870 (15)*
Belize	0.939 (15)	0.969 (33)	0.603 (149)
Colombia	*0.934 (16)*	*0.982 (20)*	0.837 (26)
Liechtenstein	0.930 (17)	1.000 (12)	0.305 (210)
Guadeloupe	0.929 (18)	0.981 (21)	0.447 (177)
New Zealand	0.929 (19)	0.975 (29)	0.751 (74)
Bahamas	0.924 (20)	0.952 (50)	0.573 (158)

Size is calculated as log area, scaled to a maximum value of 1. Rank values are shown in parentheses. Megadiversity countries (Mittermeier et al. 1997) within top 20 ranks of each column are shown in italics.

4 Conclusions

In the face of biodiversity and climate crises and their implications for future human well-being, we need to move beyond linkage diagrams and begin describing socio-economic, biological, and political attributes in conjoined and quantitative terms. Development of SBPR-based indices provides one means to advance this goal. To support such index development, the present study provided a topical review of recent literature and demonstrated a construction methodology.

The diversity of evaluation approaches identified in this review suggests a lack of consensus, within or across disciplines, on specific measures or methodologies for assessing effectiveness. Additional differences are apparent in study selection of an ex post or ex ante assessment perspective. Ex post assessment comprised over 80% of the examples provided in Table 1, while less than 50% of the distinct composite indices in Table 2 apply an ex post perspective. This discrepancy and the dearth of studies incorporating both perspectives suggest that additional research is needed to align priority setting with outcomes assessment. One means to accomplish this would be to integrate consistent data within new or existing composite indices.

While multidisciplinary studies of conservation effectiveness may remain rare within the broader conservation literature, the studies reviewed here indicate that some institutional linkages are beginning to form between disciplines. Large NGOs such as WWF and TNC have demonstrated their interest and ability to contribute toward both protected area management/establishment and economic-based incentives such as PES program development (Tallis et al. 2009). Advancements in economic evaluation and cost-effectiveness assessment have spurred further development of interdisciplinary studies. However, this review also identifies continued deficiencies in effectiveness assessment including: socio-cultural research, action on local engagement and participatory approaches, and replication across scales. Furthermore, only one in five of the indices reviewed in this study incorporate measures of biodiversity benefits to human well-being.

Demonstrating a means to integrate benefits attributes into a composite measure of biodiversity conservation effectiveness, the example indices make several contributions toward index construction methodology. As described here (and detailed in Wright 2011), these indices provide insights into attribute normalization, objective attribute weighting, and provide final values that may be mapped and compared at a politically- and economically-relevant global scale. The SBPR framework methodology is not tied to specific variables, and therefore can be adapted to assessment at smaller spatial scales.

This is only the second study found to apply DEA to a biodiversity context and the first to apply the formulation method of Ratick et al. (2009) to conservation evaluation. DEA was found to optimize index values while objectively obtaining importance weights. Furthermore, the use of assurance regions to apply weight constraints was successfully demonstrated for the first time in a biodiversity study. More consistent data, if made widely available, would allow for observation of changes in the SBPR attributes, thereby enabling performance assessment. Methodologically, the example indices require more thorough statistical analysis of both inputs and outputs, though some correlation analysis is provided by Wright (2011). The SBPR framework for composite indices also needs to be assessed other spatial scales and applied to other linked indicator sets. Due to the advantages in objective weight application and avoided masking of attributes, as identified by Ratick et al. (2009), further research on DEA application is recommended over WLC.

Additional improvements are also needed in interdisciplinary engagement and criticism from the research community and dialogue through peer-reviewed publication. Increased input from independent and empirical studies could do well to inform policy development that receives little to no public input such as the recently negotiated Access and Benefits Sharing Protocol and 2020 targets under the CBD. Additional assistance is also needed from researchers in developing measures that expand the narrow focus on species-oriented assessment to other biologically, socially, economically, and politically relevant scales.

Lastly, but critically important, index construction and evaluation should be combined with participatory research and teaching to help bridge the global and local scales at which biodiversity and its conservation are valued. Several recent studies provide relevant examples to inform this effort including diagrams and frameworks (e.g. Bottrill et al. 2008; Cowling et al. 2008), multi-criteria decision making (e.g. Regan et al. 2007), and community engagement through workshops and interviews (e.g. Farley and Costanza 2010; Petheram and Campbell 2010). Combined with adequate outreach, these efforts will enable SBPR composite indices to enhance public understanding and communication of the biodiversity crisis by encapsulating a multi-dimensional problem within an easily-communicable metric.

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