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Monitoring national conservation progress with indicators derived from global and national datasets

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ABSTRACT

Developing indicators for monitoring biodiversity, as called for by the Convention on Biological Diversity and 2020 Aichi Targets, is challenging in many countries due to data and capacity gaps. One proposed solution is to disaggregate global datasets to generate national-level indicators for countries where these values do not exist, but to date there are few examples where this approach has been systematically applied and its efficacy investigated. Using comparisons of disaggregated global data and data generated nationally for four indicators in five tropical Andean countries, we show that the two approaches can often lead to divergent values. Differences between values gathered using these two methods vary according to country and indicator, with the average differences for all countries as 26% for forest cover loss (maximum Bolivia 31%), 10% for the Red List Index (maximum Venezuela 27% for birds), 14% for protected area coverage of Key Biodiversity Areas (maximum Colombia 25%), and 67% for carbon sequestration potential (maximum Peru 102%). Most of the variations are due to methodological differences, calling into question the reliability of inter-country comparisons and roll-ups of national indicator data to regional or global scales. Nationally-generated indicators are desirable because they have the greatest power to influence national policy. However, in cases where regional or global consistency is needed, such as assessments by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services and Global Environmental Outlook, assessors should rely on global and regionally-disaggregated global data to elucidate trends and spatial patterns for most indicators. To broaden the utility of nationally-generated indicators, the biodiversity indicators community must agree on methodological standards, ensure that local stakeholders' needs are understood and addressed, develop incentives for the use of these standards, and communicate them to practitioners at all levels.

1. Introduction

Concern about the deteriorating status of biodiversity worldwide has led to the establishment of a number of policy platforms to promote responses to this crisis and chart progress toward specified targets. The Strategic Plan for Biodiversity 2011–2020 and corresponding 20 Aichi Targets (CBD, 2010), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES; Opgenoorth and Faith, 2013), the Global Environmental Outlook (GEO; UNEP, 2012), and the Sustainable Development Goals (Sachs, 2012) are four such mechanisms that either set biodiversity goals or chart societal progress in reducing declines in biodiversity. The existence of these platforms and others has created a need for the development of indicators for specified targets. Indicators are typically derived from global sources (Butchart et al., 2010; Tittensor et al., 2014), but may also be rolled up from nationally-generated sources.

Abbreviations: AZE, Alliance for Zero Extinction; CBD, Convention on Biological Diversity; DGFFS, Dirección de Gestión forestal y de Fauna Silvestre; FAN, Fundacion Amigos de la Naturaleza, Bolivia; IBA, Important Bird Areas; IPBES, Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services; IUCN, The International Union for Conservation of Nature; GEO, Global Environmental Outlook; KBA, Key Biodiversity Areas; MAE, Ministerio del Ambiente del Ecuador; MINAM, Ministerio del Ambiente del Peru; MINAG, Ministerio de Agricultura, Peru; MMAA, Ministerio de Medio Ambiente y Agua, Bolivia; NBSAP, National Biodiversity Strategic Action Plans; RLI, Red List Index; REDD, Reduced Emissions from Deforestation and Forest Degradation; RUNAP, Registro Unico Nacional de Areas Protegidas, Colombia; SERNANP, Servicio Nacional de Areas Naturales Protegidas, Peru; SERNAP, Servicio Nacional de Areas Protegidas, Bolivia; WDPA, World Database on Protected Areas.

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Nationally-generated data play a necessary role in this arena because they address biodiversity issues at a scale relevant to the governments charged with improving the status of biodiversity (Soberon and Sarukhan, 2009; Stephenson et al., 2015). Classification of remotely-sensed data by local technicians familiar with the land forms depicted in the imagery can be more accurate than global classification schemes that lump features into broad categories (e.g., UNEP-WCMC, 2015). Similarly, assessment of population status by field biologists familiar with species in a particular country can provide a more accurate representation of species' extinction risk in that country than estimates made across the ranges of the species, many of which typically include multiple countries where threats can vary substantially. Furthermore, the resolution of mapped data can be finer for nationally-generated indicators than for global indicators derived from relatively coarse resolution global data. Finally, a key value of indicators compiled from nationally-generated data is that they speak directly to targets set by national level stakeholders, the same entities that are often responsible for maintaining healthy ecosystems. Through their contributions to the identification of both targets and indicators for measuring progress, these stakeholders are more invested in achieving positive outcomes (Soberon and Sarukhan, 2009). By contrast, no single entity is responsible for achieving global targets, which are set via multilateral processes.

Despite the advantages of nationally-generated indicator data, the availability of these data and the capacity and willingness to generate indicator data vary among countries (Bubb, 2013; Han et al., 2014). To fill gaps in nationally-generated data, the conservation community has proposed disaggregating global datasets at the national level to use as a bridge until countries develop their own capacity to compile data and derive indicators (Bowles-Newark et al., 2015a; Bubb, 2013; Secades et al., 2014; Stephenson et al., 2015). The resulting indicators could be used, for example, in National Biodiversity Strategic Action Plans (NBSAPs), which are key to implementing the CBD at the national level. However, to date there have been few examples where this approach has been systematically applied, and its efficacy is untested. Here we explore the concordance of a sample of disaggregated global and national indicators that track different aspects of biodiversity. We ask whether the indicators tell the same story, what factors might cause differences, and in which situations each data source might be more powerful. The answers to these questions provide important input to determining strategies for future indicator development and use.

2. Material and methods

2.1. Biodiversity indicators from tropical Andean countries

We selected five tropical Andean countries (Fig. 1) for our comparison of globally disaggregated and nationally-generated indicators for two reasons. First, the tropical Andes is representative of many tropical regions that harbor exceptionally high biodiversity and are confronting urgent threats. Second, a previous study indicated that monitoring capacity is higher in tropical Andean countries than in some other tropical regions of Africa and Asia (Han et al., 2014) and thus nationally-generated data should be more readily available and for a longer time series.

We chose four indicators for comparison, one each from the pressure-state-response-benefit framework used by the CBD (Bubb et al., 2011; Sparks et al., 2011; UNEP-WCMC, 2009) (Table 1). These indicators are largely consistent with those presented via the Biodiversity Indicators Dashboard (BID; http://dashboard.natureserve.org) and are highly relevant to global biodiversity monitoring initiatives, including the 2020 Aichi targets. The availability of national data varied by indicator; we sought to compare as many countries as possible for any given indicator, and in one instance needed to restrict the assessment area to sub-national units to be consistent with data availability. The small sample size, both in terms of the number of countries with data for any given indicator and the number of indicators for which comparison of global and nationally derived values was possible, is indicative of the general difficulty of obtaining comparable metrics between countries. This small sample size precludes statistically robust comparisons of differences in nationally and globally derived biodiversity indicators, yet our results still provide a compelling means to illustrate issues that arise when applying data from these disparate sources.

2.1.1. Forest cover loss

We calculated the annual loss of forest cover as an indicator of the rate of deforestation. For both the global and national indicators, this value represents the annual loss in forests as a percent of the year 2000 forest cover baseline. The disaggregated global values were derived from the Global Forest Change dataset (Hansen et al., 2013), which mapped global forest tree cover and its change from 2000 to 2012 using Landsat imagery at 30-m spatial resolution. Data sufficient for calculating national forest loss values have been produced and published for Bolivia, Colombia, Ecuador, and Peru, but not Venezuela (Table 1). These values were derived from national forest cover maps developed using satellite imagery (primarily Landsat, but also ASTER; see the Supplemental materials Appendix for more information).

National data on forest cover differed among countries and from the global data in the minimum mapping units used, how forests were defined, the specific classification techniques employed, and the dates for which data were available (see the Supplemental materials Appendix for a complete summary). Whereas the Global Forest Change data did not consider a minimum mapping unit, each national estimate did; the areas ranged from a low of 0.3 ha for Bolivia to a high of 25 ha for Colombia. Forests were defined based on percent canopy cover and canopy height. The Global Forest Change data mapped tree cover, and we defined forests as 30-m pixels with at least 25% cover of trees at least 5 m high following Hansen et al. (2010), whereas nationally, canopy cover requirements ranged from 10% (Peru) to 30% (Ecuador) and canopy height requirements ranged from 3 m (Peru) to 5 m (all other countries for which canopy height was specified). Forest plantations were included in the definition of forest for the global data, Ecuador and Peru, but not for Colombia (whether they were included in Bolivia is not clear). Regenerating forest was considered as forest in the global and Ecuadorian estimates, but the method descriptions for the remaining countries do not address this point. For Bolivia, Colombia, and Peru, national data were available for 2000 and 2010, and the national and global indicators of forest loss reflect change between these dates. For Ecuador, national data were available only for 2000 and 2008; the national and global indicators presented for Ecuador reflect forest loss between these years.

2.1.2. Red List Index (RLI)

The Red List Index is a measure of trends in survival probability (the inverse of extinction risk) for sets of species within broad taxonomic groups. It is based on the numbers of species within each IUCN Red List category and the changes in these numbers over time resulting from genuine improvement or deterioration in status between assessments (Butchart et al., 2004, 2005, 2007, 2010; Hoffmann et al., 2010; IUCN, 2010). This standardized RLI varies between 1 (all species Least Concern) and 0 (all species Extinct or Extinct in the Wild). We sought to compare RLI results from national assessments with country-specific results derived from comprehensive global assessments, contrasting both the most recently calculated RLIs by taxonomic group for each country, and, where assessments from multiple years were available, calculating the annual change in aggregate extinction risk by dividing the difference in RLI from the last to first assessment by the number of intervening years.

For the disaggregated global value of this indicator, we used the last comprehensive Red List assessment for each of three vertebrate groups for the RLI, and first and last comprehensive assessments to calculate annual change in RLI (1988 and 2008 for birds, 1996 and 2008 for mammals, and 1980 and 2004 for amphibians) following Butchart et al.

Fig. 1. Map of northwestern South America showing tropical Andean countries included in study. The hashed lines delimit the Amazon Basin in Colombia and Peru where the benefit indicator (carbon sequestration potential) was measured.

(2004, 2005, 2007 and 2010) and Rodrigues et al. (2014). Only the species undergoing genuine changes were included using the Rodrigues et al. (2014) dataset. We identified all species falling partially or completely within each country using 2010 spatial distribution data for each species (IUCN, 2010), and weighted them based on the proportion of the distributional range in the country (Rodrigues et al., 2014).

We compared the disaggregated global RLI with RLI values calculated from published comprehensive national Red List assessments for Bolivia, Ecuador, and Venezuela (Table 1) (methods follow Butchart et al., 2007; Butchart et al., 2004; Collen et al., 2013). The national Red List assessments for Ecuador (1996 and 2011 for mammals) and Venezuela (1999 and 2008 for amphibians, birds and mammals) followed the

Table 1
Sources for indicators derived from disaggregated global and national datasets.

Indicator category	Indicator	Global data source	Countries with data	National data source
Pressure	Annual loss of forest cover	Global forest change from remote sensing data (Hansen et al., 2013)	Bolivia, Colombia, Ecuador, and Peru	National reports or national land use-land cover maps based on remote sensing data (Cabrera et al., 2011; Che Piu and Menton, 2013; MINAG and DGFFS, 2010; MINAM, 2009; MINAM, 2011a; MAE, 2013; SERNAP, 2013)
State	Red List Index	IUCN Red List of threatened species (Han et al., 2014; Rodrigues et al., 2014)	Bolivia, Ecuador, and Venezuela	National Red List assessments (Aguayo, 2009; Balderrama, 2009; MMAA, 2009; ProVita, 2014; Tarifa and Aguirre, 2009; Tirira, 2001, 2011; UICN-Sur et al., 1997)
Response	Protected area of key biodiversity areas	World Database on Protected Areas (BirdLife International, 2014; IUCN and UNEP, 2015)	Bolivia, Colombia, Ecuador, Peru, and Venezuela	Protected area boundaries from national authorities (FAN, 2009; MINAM, 2011b; ProVita, 2005; RUNAP, 2013; SERNANP, 2013).
Benefit	Potential for carbon sequestration (Colombian and Peruvian Amazon basins)	Forest carbon map (Saatchi et al., 2011) and global forest change data (Hansen et al., 2013)	Colombia, Peru	Colombian and Peruvian national reports on carbon stock (Asner et al., 2014a; Asner et al., 2014b; Phillips et al., 2011) and forest cover change (Llactayo et al., 2013; Murcia et al., 2010; Murcia et al., 2013).

International Union for the Conservation of Nature (IUCN) guidelines for regional assessments (IUCN, 2012a or previous versions). The Bolivian assessments (2008 for amphibians, birds and mammals) used an independent methodology that combined information on distribution, conservation status of the species' habitat, population status, intrinsic vulnerability, and threats, but assigned the same category names as the IUCN. National Red List assessments that follow the IUCN criteria assign categories to species depending on their risk of extirpation within a country, ignoring extra-limital populations except to the extent that these populations can be the source of individuals dispersing into countries to "rescue" extirpated populations (Collen et al., 2013). Because the focus of national Red Lists is exclusive to the country being assessed, a national RLI weighs each species equally (as in the global RLI). We excluded national Red List assessments that did not attempt to comprehensively assess all species in a taxonomic group. For this reason we do not report results for Colombia or Peru where national assessments do not comprehensively cover different taxonomic groups. To calculate the annual change in aggregate extinction risk for the national data, we enlisted the assessment authors to back-cast the Red List status of Ecuadorian mammals that were not originally assessed in 1996 and Venezuelan species that were not originally assessed in 1999 (Butchart et al., 2007). The genuine changes were systematically identified by the assessment authors. We were unable to obtain data from multiple years from Bolivia.

2.1.3. Protected area coverage of key biodiversity areas (KBAs)

This indicator reflects the degree to which KBAs, sites identified as contributing significantly to the global persistence of biodiversity (Eken et al., 2004), fall within protected area boundaries for each country (Butchart et al., 2012). The mean percent area covered by protected areas for all KBAs within each country was calculated via spatial overlay, first based on protected areas as defined in the WDPA (IUCN and UNEP, 2015), and again using spatial data on protected areas maintained by individual countries (Table 1). The 630 KBAs identified in tropical Andean countries are comprised of 538 Important Bird & Biodiversity Areas (BirdLife International, 2014), 127 Alliance for Zero Extinction sites (areas holding effectively the entire populations of highly threatened species; Ricketts et al., 2005), and six additional sites that meet the KBA definition (Young et al., 2015). Overlapping KBA areas were counted only once.

2.1.4. Carbon sequestration potential

Carbon sequestration potential is the product of carbon biomass and deforestation rate (Kindermann et al., 2008) and is derived using the same formula for globally disaggregated and national indicators, but with different sources of data for both carbon biomass and deforestation rates. For both indicators, we calculated a carbon sequestration value for each country by multiplying carbon stock by the mean annual deforestation rates and a conversion factor of 3.66 CO_2 equivalents to account for the higher molecular weight of CO_2 compared to C.

For the disaggregated global values, total aboveground carbon biomass was obtained using a global forest map of carbon stock for tropical areas (Saatchi et al., 2011). Annual deforestation rates were derived from global forest change data (Hansen et al., 2013) using methods consistent with those used for the forest loss indicator but defining forests as areas with at least 10% cover of trees at least 5 m height (to match measures used in national data sets). Comparable national data were only available for Colombia. Peru also had national data, but forest loss rates for regions outside of the Amazon Basin had high uncertainty. To be able to include calculations with low uncertainty and with both Colombia and Peru in the comparison, we report carbon sequestration potential for only the Amazon Basin of these two countries. Carbon biomass (Asner et al., 2014b; Phillips et al., 2011) and forest change (Llactayo et al., 2013; Murcia et al., 2013) data were obtained from national authorities.

More methodological details are found in the Supplemental on-line Appendix.

3. Results and discussion

3.1. Indicator concordance

The degree of indicator concordance between nationally-derived and globally disaggregated data differed both among indicators, and between countries for the same indicator. Variation in indicator values is reported as the percent difference between the globally disaggregated and nationally-derived values.

3.1.1. Forest cover loss

The patterns of forest cover loss were broadly similar for disaggregated global and nationally-reported estimates of annual forest cover loss (Fig. 2). Both global and national indicators agree on a relatively low deforestation rate in Peru, but global data point to Bolivia as having the highest deforestation rate whereas national data highlight Ecuador and Colombia (Fig. 2). The magnitude of the differences ranged from 0.04 to 0.12% of forest cover lost annually, with the percent difference in indicator values varying from 22% (Peru) to 32% (Bolivia). To put these values in perspective, the disaggregated global value estimates nearly 21,000 km^2 more loss of forest cover over ten years than the Bolivian national estimate.

The sources of disagreement most likely stem from methodological differences. Although Landsat imagery was central to all global and national estimates, differences in the amount of canopy cover, tree height, and minimum block areas required to designate forest likely impacted the estimate. In addition, plantations and regenerated forest were mapped as forest area in some country sources but not others. Finally, different algorithms, supplemental imagery and other supplemental data were used in classification processes for training data, ground truthing approaches, and methods for assessing areas where cloud cover obscured Landsat images also likely contributed to differences in the estimates. The methods used in Peru are as divergent from those used in the global estimate as for any other country in the sample, and thus no obvious methodological congruency can explain the similarity between the Peruvian and global values.

3.1.2. Red List Index

Comprehensive national Red List assessments are available for birds, mammals, and amphibians in a subset of tropical Andean countries (Fig. 3). The magnitude of the differences in the RLI between global and national assessments ranges from 0.03 to 0.20, with a percent difference of 4–27%. The disaggregated global RLI estimate points to a lower extinction risk for birds and mammals but a higher extinction risk for

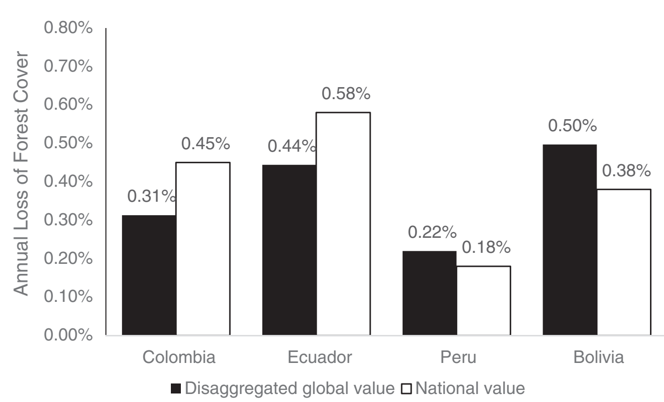

Fig. 2. Comparison of disaggregated global and national data values for annual loss of forest cover of tropical Andean countries for the period 2000–2010, except for Ecuador where the estimate is for 2000–2008.

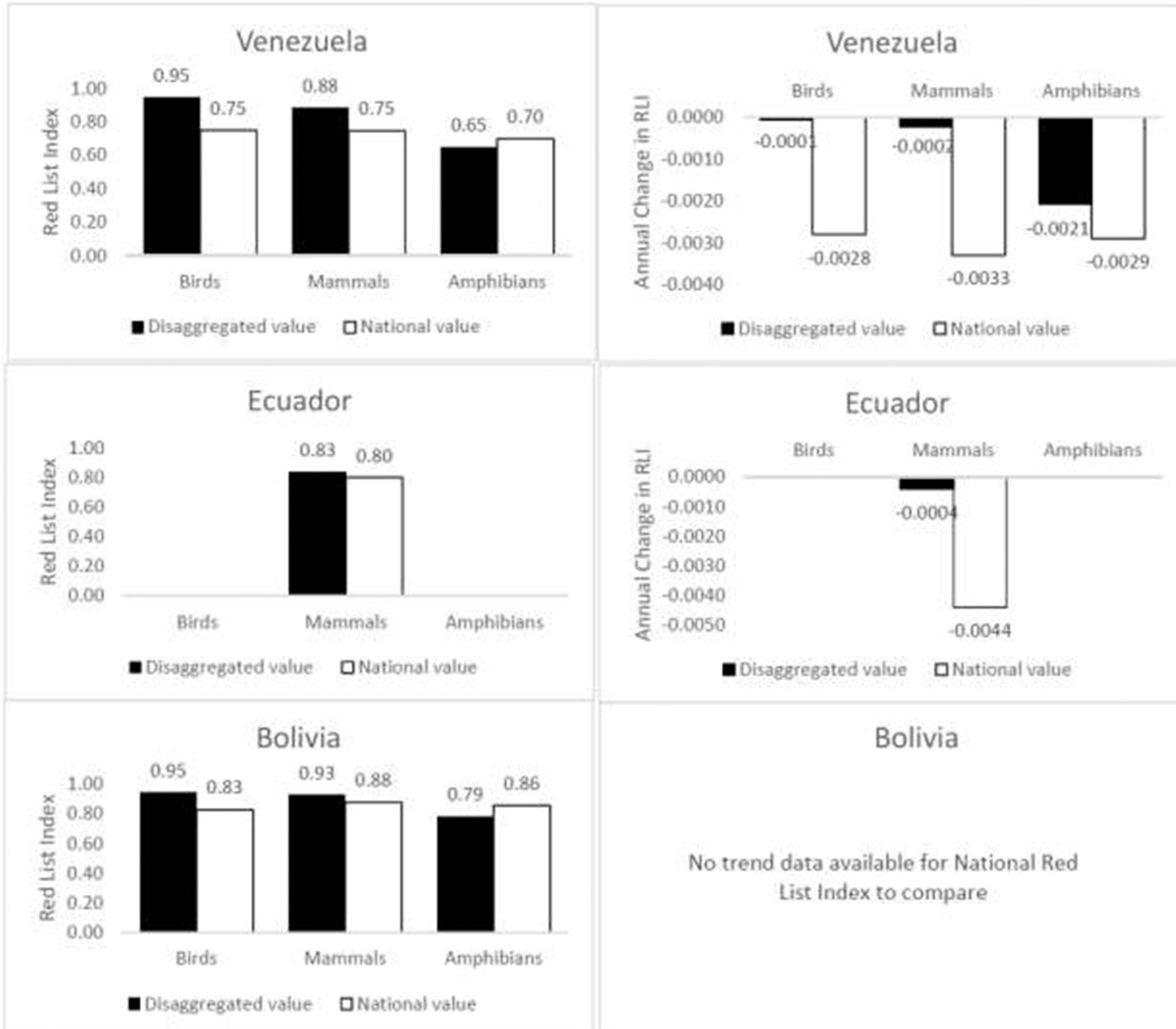

Fig. 3. Comparison of disaggregated global and nationally-reported values for the Red List Index (left) and annual change of Red List Index (RLI; right) for tropical Andean countries where data are available.

amphibians as compared to the RLIs calculated from national assessments. In the four cases where a rate of change in RLI can be calculated, the disaggregated global value always resulted in a lower estimate of the rate of decline toward extinction as compared with the national assessments (Fig. 3).

In theory, the smaller ranges examined in national assessments (for non-endemic species; range sizes for endemics are identical) compared to global assessments should result in higher threat categories because a measure of range size is one of the criteria considered in assigning Red List categories (IUCN, 2012). Thus the higher extinction risk found in the national assessments is expected. The amphibian assessments do not follow this pattern, however. An explanation may be that the global assessments were performed in the early 2000s at the height of amphibian declines that took place in the tropical Andes (Ron et al., 2003; Stuart et al., 2004). Since then, new populations for some species have been discovered (e.g., Rojas-Runjaic et al., 2014), perhaps leading to assessment in lower risk categories in national assessments that have taken place more recently (i.e., in 2008) than the global assessment (i.e., in 2004). One driver of the more rapid increase in extinction risk shown in the national assessments might be lower extinction risk assignments in the global data for the many wide-ranging bird and mammal species whose distributions extend across Amazonia Brazil, where threats to those species are less widespread. However, to some degree this would have been offset by the fact that the disaggregated global assessment weighs species by the proportion of their distribution in the country. Furthermore, differences in judgments about which changes in Red List categories were a result of genuine changes in extinction risk may explain the lower rates of change for the global indices as well. Globally, compilers of Red List species are cautious about claiming a genuine change unless there is reasonable supporting evidence.

3.1.3. Protected area coverage of KBAs

Using disaggregated global versus nationally-provided data on protected area revealed a fairly similar pattern of the degree to which KBAs are covered by protected areas (Fig. 4). The two measures coincided in the rank order of the first through third countries with the highest portions of KBAs covered by protected areas, but reversed the sequence of the fourth and fifth countries. Nevertheless, the percent difference in indicator values varied by up to 25% (Peru), with an absolute value of the mean proportion of KBAs protected as high as 7% in Peru and 8% in Colombia.

Because the KBA outlines are constant, the differences between the two data sets are entirely due to differences in protected area boundaries. Considering that protected areas are continually being declared, one might predict that the global database, which takes some time to

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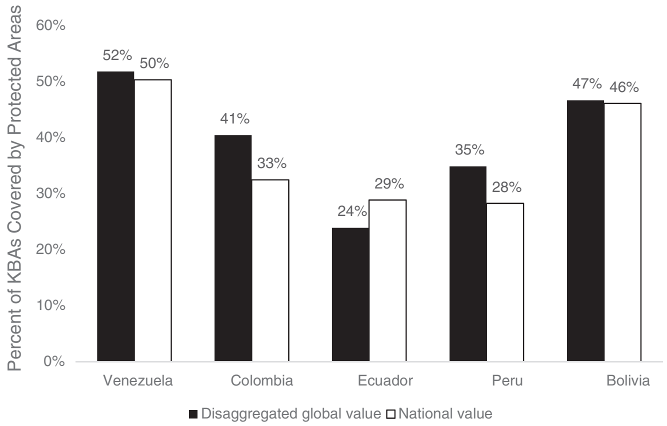

Fig. 4. Protected area coverage of key biodiversity areas in tropical Andean countries, calculated using disaggregated global and nationally-reported protected area data.

be updated after new protected areas are established, would be less complete and therefore under-predict coverage of KBAs. Indeed, the World Database of Protected Areas misses the recently established Ecuadorian protected areas Colonso Chalupas, Galera San Francisco, Siete Iglesias, El Zarza, and Cofan Bermejo, as well as the recent expansion of Sangay National Park. However, the global dataset returned a higher estimate of coverage of KBAs in four of the five countries examined. Although degazetting and downsizing of protected areas has occurred in some countries (and this may not yet have been reflected in the WDPA), this phenomenon has been most extensively reported in Ecuador, the one country where national data show a greater coverage of KBAs than the global data (Mascia and Pailler, 2011).

Major sources of the discrepancies are caused by differences in the age of the data and how the WDPA and national data sources define the objectives of land use categories. The WDPA is a compilation of national data, prioritizing government-validated data over NGO data, into a global database. However, national governments do not always provide timely updates to the WDPA. In particular, the WDPA data for Venezuela are somewhat dated. In addition, differences in interpretation of the objectives of public lands can result in the WDPA tending to over-represent the number and extent of protected areas relative to national reporting (Fig. 5). Two examples support this contention. First, Venezuelan "forest reserves" are defined as protected areas by the WDPA but not by the government, which considers that these lands are not protected because they are subject to legal logging. Second, the WDPA lists protected area buffer zones in Peru as protected whereas the national database does not.

3.1.4. Carbon sequestration potential

Of the four indicators examined, we found the greatest differences in values derived from global versus national data in carbon sequestration

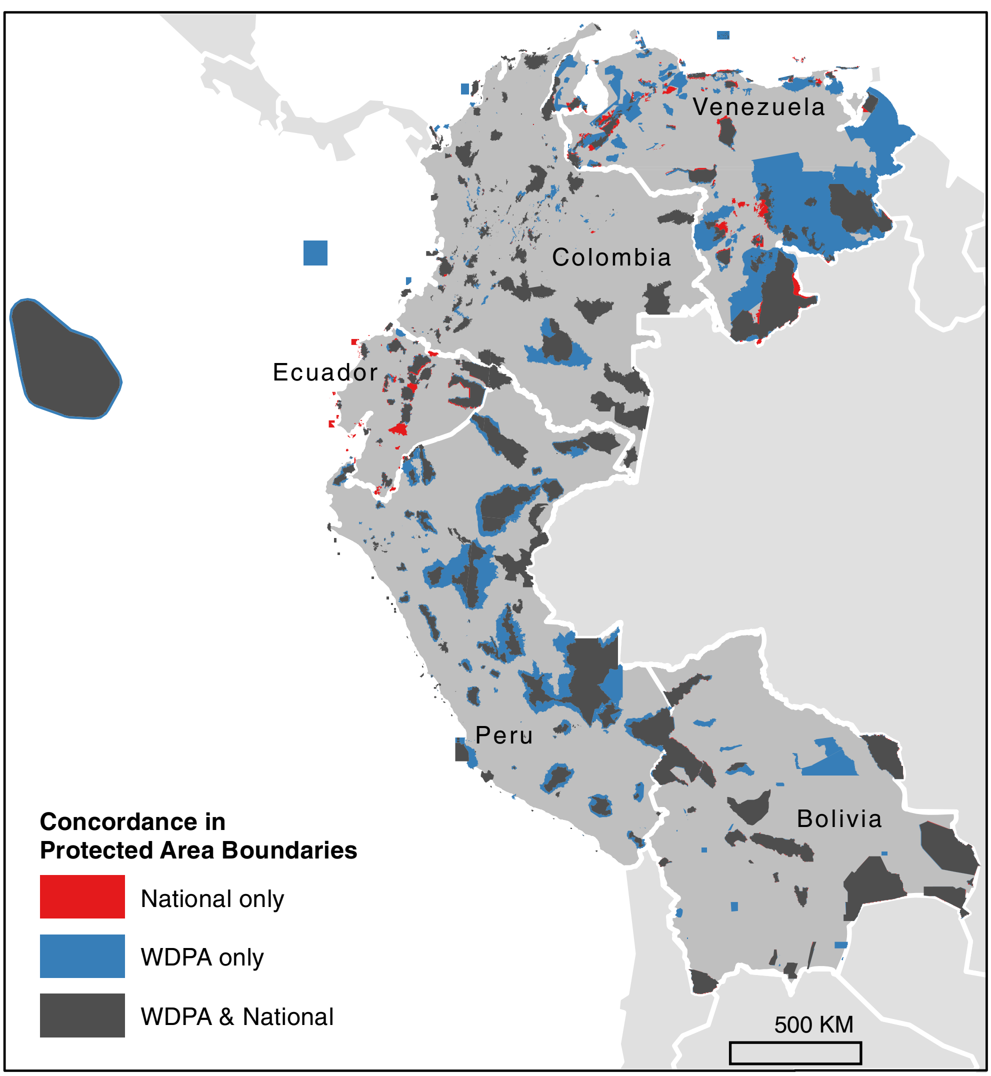

Fig. 5. Differences in protected area boundaries between global (World Database on Protected Areas, WDPA) and national data sources in tropical Andean countries.

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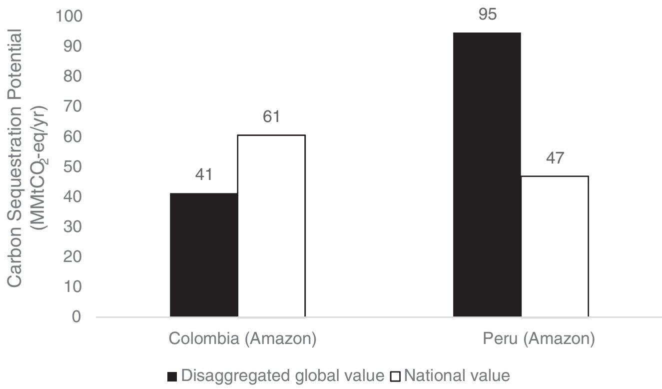

Fig. 6. Comparison of estimates of carbon sequestration potential (million metric tons of CO_2 equivalents per year) in the Amazon Basin of Colombia and Peru using disaggregated global and national estimates of carbon biomass and deforestation rates.

potential. Whereas the calculation using global data was 49% lower than the estimate using national data in the Amazon Basin of Colombia, this estimate was 102% higher in the Peruvian Amazon (Fig. 6). These differences were caused by differences in the underlying national and global values for deforestation and carbon stock in the two countries. In Peru, the national and global values for deforestation in the Amazon Basin were similar (0.1% and 0.2%, respectively), but the carbon stock estimate was much lower in the national than global estimate (6.5 and 19.7 billion MgC, respectively). Conversely, in Colombia the carbon stock values were similar (5.2 billion MgC national, 7.7 billion MgC global) but the national deforestation estimate was much higher than the global value (3.2% versus 1.4%, respectively). Differences in forest biomass estimates may result from the varying methods used to calculate this measure: MODIS, STRM, and plot data (global estimate); plot data and extrapolation (Colombia); and LiDAR imagery (Peru).

3.2. Implications for indicator development and use

Our findings demonstrate that major indicators of biodiversity at the national scale can vary substantially depending on whether they are derived from disaggregating global data sets or measured through a national process, and that there appears to be little pattern to the variations. Defensible differences in methodologies can lead to variations in indicator values from disparate sources, which are not indicative of any actual differences in biodiversity pressures, states, responses, or benefits. Where available, nationally-derived indicators are often most relevant in measuring biodiversity status and trends for the countries where they are developed. Disaggregated global data becomes important for filling in the gaps in nationally-derived data or providing consistency in measures across international boundaries (Stephenson et al., 2015). Our results underscore the importance of understanding how data from those sources may differ for any particular conservation question, as well as the essential role of on-going efforts to increase the standardization of indicators across boundaries.

As more indicators are derived from both global and national sources in the future, statistical analyses characterizing the discrepancies will become feasible. The more evidence-based indicators used in updates of national reports on progress toward the Aichi Targets, promotion of indicator development at both national and global levels through international such as the Biodiversity Indicators Partnership (BIP; http://www.bipindicators.net/) and the United Nations Framework Convention on Climate Change (UNFCCC; http://unfccc.int/2860.php), and the visualization and communication of these indicators through platforms such as the Biodiversity Indicators Dashboard (BID; Han et al., 2014), should catalyze the emergence of many more indicator sets that would support statistical analyses. Such an effort, if extended to include non-Andean countries, would address the generality of the results reported here for a small subset of countries.

The causes of variation in indicator values are often easy to understand, yet hard to fully quantify. A lack of consensus on standards for measurement and monitoring can lead to disparate results. For example, the indicators with the largest observed percent differences were those for which methods continue to be debated, including approaches to monitoring forest carbon, a topic that has important policy implications for both negotiations on the REDD (Reduced Emissions from Deforestation and Forest Degradation) mechanism as well as on calculations of global carbon emissions and effects on climate (e.g., Goetz and Dubayah, 2011; Le Toan et al., 2011; Taylor et al., 2015). A secondary cause of variation in indicator measurement is intrinsic to the smaller spatial scale of national compared to global assessments, as the Red List assessments demonstrated. In some cases, such as amphibian RLI estimates, we can only speculate about the cause of the difference between global and national indicator values.

The variation in indicator values that we revealed underscores that the conservation community must exercise care when deploying national indicators in situations where consistency across international boundaries is a compelling need. International processes such as the Aichi Biodiversity Targets, GEO, and IPBES focus on global outcomes, informed in the case of GEO and IPBES by regional assessments. In these cases, methodological consistency is key to supporting the rigor of the assessments. Disaggregated global data may prove useful in this situation, especially because the data are needed at regional and not national scales (Brooks et al., 2016). Efforts to roll up nationally reported data to characterize global status or trends will need to pay attention to potential methodological inconsistencies, and should be undertaken with the knowledge that resolving national differences in reporting is likely to be a significant challenge.

Failing to explain sources of variation between globally and nationally derived indicators for global assessments, or defaulting to globally-derived data for national applications, may at best cause confusion in countries that have developed data for the same indicators and at worst lead to questions of data credibility (Han et al., 2014). Disaggregated global indicators can be most effectively used in cases where these data have not been generated at national levels. Having some estimate of the rate of forest loss, a fundamental measure of threat to biodiversity in biomes with forest cover, in countries that do not report this figure is valuable for many reasons, including directing international conservation efforts. Similarly, having RLI data for countries without national Red Listing processes is useful to pointing out where species are sliding most rapidly toward extinction. However, nationally-generated indicators may be better positioned to respond to local needs, be replicated over time, and inform local policy decisions (Soberon and Sarukhan, 2009) and should thus be used, where available, to address conservation concerns at the national level. The national Red List of Ecuadorian mammals (Tirira, 2001, 2011), which has now been conducted three times, is a good example of how local initiatives can lead to valuable indicators.

Despite choosing a focal region known to have relatively high capacity to generate indicator data, and focusing on indicators for which data was likely to be available, we found that gaps in data availability are persistent. National data for every country in our study area were available for only one of the four indicators examined, i.e., the protected area coverage of KBAs. Fortunately, the international community is invested in supporting increased local capacity to monitor trends in biodiversity. For example, one of the four foundations of the IPBES workplan is to enhance the capacity of nations to be able to carry out biodiversity assessments (Bowles-Newark et al., 2015b; Opgenoorth and Faith, 2013). In some cases, such as for Guyana, international environmental organizations are contributing expertise to the development of NBSAPs and the indicators reported therein (Bowles-Newark et al., 2015b). In addition, remote sensing technology continues to improve and remote sensed data are becoming more widely available. As the sophistication of the

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national analyses of forest loss rate demonstrates, local capacity to analyze remote sensed data also continues to improve. The result is an expansion of the data available to assess trends in biodiversity and derive finer resolution products, resulting in the potential for better scientific input into policy decisions (Secades et al., 2014). Certainly the increased volume of information generated by governments and civil society will expand our understanding of processes that affect biodiversity and increase our ability to detect trends in these measures. However, our findings suggest that methodological differences are likely to emerge when different countries set out to measure similar variables.

Entities responsible for generating the data behind indicators should be cognizant of the importance of documenting clearly and completely the methods followed to generate data products, and to the extent possible follow international standards. For example, although our study examined just four national reports of forest cover loss, we uncovered four different ways to define a forested area. Moreover, several aspects of the definition of forests, such as whether regenerating forest was included or what ground truthing methods were used, was unclear from the descriptions of the studies providing forest cover loss estimates.

International efforts that promote standards for monitoring data (and incentives for following them) can maximize the degree to which increases in national capacity for monitoring result in an increased availability and comparability of indicators across countries. A key step toward achieving this is the early identification and engagement of a wide range of stakeholders, including local communities, to increase the utility of indicators and data (Brown et al., 2014; Danielsen et al., 2014). Integrating an ongoing dialogue with end users across scales into the design phase of data collection and mobilization projects facilitates the interpretation of findings and can improve policy relevance.

Involving local stakeholders in the development of standards can enhance the likelihood that the standards are adhered to. A good example is the revision of the KBA standard, now nearing completion, which involved a worldwide consultative process (Brooks et al., 2015). Standards have emerged for Red Lists of species and ecosystems as well as other data sets that underpin indicators for numerous Aichi Targets (Brooks et al., 2015). A major remaining task is reaching an agreement on how to define forests for forest cover loss measurements derived from remotely-sensed data. Such a standard may require flexibility to accommodate differences in forest structure across biomes and characteristics of satellite-derived data products. A regional process to develop such standards has been initiated by the European National Forest Inventory Network (Vidal et al., 2016), but needs further development for countries outside of Europe. Additional efforts to establish standards for other indicators, as well as communication with practitioners to ensure that the standards fit their needs, would substantially increase comparability of nationally-derived indicators and concordance with disaggregated global values.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found in the online version, at http://dx.doi.org/10.1016/j.biocon.2016.08.023. These data include Google maps of the most important areas described in this article.

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