Better practices for reporting on conservation
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Abstract
Trend indicators are the primary approach used for reporting on biodiversity worldwide, but often poorly inform conservation policy and management. Here, we show how the field of systematic conservation planning offers approaches for biodiversity reporting to foster better adaptation and accountability by estimating the difference made by conservation interventions; identifying how changes in biodiversity contribute to conservation goals, accounting for unequal and complementary contributions; and evaluating cost-effectiveness of interventions. By recognizing that biodiversity reporting and conservation prioritization must inform each other as an adaptive process, we show how they share data needs and methodologies, including distributions and abundances of features and pressures, predictions of future changes in features under different pressures, distributions of different interventions and their associated costs, and stepwise models aggregating contributions to an overall goal. Incorporating prioritization-based approaches into biodiversity reporting will enable more robust conservation decisions than would be possible based on simple trend indicators.

Introduction
Governments and independent organizations worldwide are establishing reporting systems that collect and present information on biodiversity to stakeholders. The hope is that these systems will aid conservation decision-making and adherence to international agreements, such as the Aichi Biodiversity Targets. The primary approach to biodiversity reporting internationally is to use trend indicators that track changes in individual or multiple features of biodiversity over time (Butchart et al. 2010; Jones et al. 2011). Simple trend indicators can be readily communicated to stakeholders (Balmford et al. 2005; Jones et al. 2011; Pereira et al. 2013), allowing them to attract public attention, galvanize public discourse, and set policy agendas (Hammond et al. 1995). Some also claim that trend indicators can assist policy decisions and measure how well policies are working. For example, changes in the population sizes of birds have been used to track biodiversity impacts associated with agricultural policies (Balmford et al. 2005). Indeed, ability to connect with policy has been a key rationale for the development of many biodiversity trend indicators (Walpole et al. 2009; Nicholson et al. 2012).

Most biodiversity trend indicators, however, have three important deficiencies when applied to assisting conservation policy and management decisions and measuring how well policies work. First, and most fundamentally, few trend indicators are designed to link changes in biodiversity features to the pressures affecting them and the conservation interventions attempting to mitigate those pressures (Nichols & Williams 2006; Gardner 2010). This compromises their ability to infer policy effectiveness (Nicholson et al. 2012), and learn from management decisions in order to adapt future actions (Nichols & Williams 2006). Second, interpretation of trend indicators is hampered by a lack of clarity about how changes in the features that they track relate to higher-level conservation goals, such as the overall state of biodiversity. Third, these two deficiencies preclude considerations of cost-effectiveness in allocating scarce conservation resources.
Table 1 Measurements needed to enable biodiversity reporting to complement and inform conservation prioritization (CP). Each attribute has one or more associated data needs

| What to measure | Relevant CP concepts | Data required |
|-----------------|----------------------|---------------|
| Responses of features to pressures, and changes in pressures (and hence features) caused by policy and management interventions | Vulnerability | Distributions and abundances of pressures, Distributions of interventions, Shapes of relationships between biodiversity features and pressures, and between pressures and interventions (e.g., exponential, power, quadratic, etc.) |
| Contributions of changes in features to goals or the overall state of biodiversity in different contexts | Irreplaceability, complementarity | Past and present distributions and abundances of biodiversity features; Methods of aggregating context-dependent contributions to intervention costs |
| Cost-effectiveness of different policy and management interventions | Cost-effectiveness | |

Nicholson et al. (2012) recognized that biodiversity data collection needs to be “…set within a framework that enables informed decisions to be made, and their impact on biodiversity evaluated both a priori and through ongoing monitoring.” We agree, and propose that the field of conservation prioritization can provide the necessary framework to inform the design of biodiversity information gathering and reporting schemes in a way that strengthens connections between reporting and decision-making. Here, we use the term conservation prioritization to mean the decisions that attend the allocation of resources among diverse opportunities for achieving conservation goals. Many of the conceptual and practical tools of conservation prioritization are generalized from the field of systematic conservation planning, which was originally devised to design protected areas networks that maximized the representativeness and persistence of features of biodiversity, such as species, within a region (Margules & Pressey 2000; Moilanen et al. 2009).

The challenges of conservation prioritization have inspired three conceptual and practical developments relevant to reporting (Table 1). First, prioritization requires articulation of the difference that will be made by conservation interventions, considering expected changes to biodiversity features in their absence. Second, it is recognized that the contribution of a feature to a conservation goal is context-dependent, and must be calculated afresh with each addition or loss to a suite of features such as species or sites. Conservation prioritization therefore uses stepwise (i.e., iterative) methods for scaling and aggregating contributions to an overall goal as progress toward it changes. Third, the ability to estimate difference made in different contexts also allows prioritization to evaluate the cost-effectiveness of potential conservation interventions.

Here, we suggest that applying developments in conservation prioritization to biodiversity reporting would better inform policy and management responses. We argue that this is because linking biodiversity reporting to conservation prioritization provides much more direct information on the features that are targeted by management. Although it requires data exceeding that needed to report on trends in indicators, our proposed approach to biodiversity assessment has the advantage of actively learning from policy and management in order to show the difference made to biodiversity by different conservation interventions. Active adaptive management has only been seen as essential to the planning of future interventions (Grantham et al. 2009), and not reporting. We conclude that as biodiversity declines accelerate and governments worldwide respond, there is an urgent need to view biodiversity reporting and conservation action as reciprocal activities, and to design a more logical system for gathering the data essential to making sound conservation decisions.

Measuring “difference made”

Designing and adapting sound conservation policies requires knowledge of how much difference (if any) is made to biodiversity by relevant pressures and conservation responses. But how biodiversity responds to pressures and actions cannot be deduced retrospectively from time series data (Platt 1964), such as state-pressure-response indicators. Although time series may show declines in biodiversity state and concurrent increases in pressures or responses (e.g., Butchart et al. 2010), they do not reveal how much those trends are actually linked (see Box 1).
Box 1. Illustration of how estimates of “business-as-usual” (trends without policy) are needed to reveal the difference made by policies and management

Trends in indicators by themselves fail to provide the information about the difference made by policy or management that is needed to inform conservation. This is shown below:

**Scenario 1: Land clearance (conservation policy)**

![Graph showing percent of original habitat area remaining with and without policy, illustrating the difference made by the policy.]

In scenario 1, panel (a), trends report a decrease in the percent of original habitat area remaining, but panel (b) shows that a land clearance policy has made a positive difference by preventing some of the habitat loss that would have occurred without the policy.

**Scenario 2: Predator control (conservation management [mgmt])**

![Graph showing number of individuals with and without predator control, illustrating the difference made by the management.]

In scenario 2, panel (c), a positive trend in numbers of individuals of a species reflects only part of the difference made by predator control. Panel (d) shows that predator control added new individuals to the starting population and prevented deaths that would have arisen in the absence of management, resulting in a much larger difference made to conservation.

Conservation prioritization approaches are not founded on trends, but instead choose sets of actions (e.g., reservation of areas, predator control at certain locations) that will collectively maximize progress toward a biodiversity goal by reducing relevant pressures (Stephens et al. 2002). A forecast of the difference made to biodiversity by an intervention is calculated by subtracting expected change in the intervention’s absence (“business-as-usual,” which is the trend without policy or management [scenario B and D in Box 1]), from the change expected with its implementation (Stephens et al. 2002). Business-as-usual is usually a negative trend because there is expected loss or vulnerability (see Box 2 for a list of key terms).
**Box 2. Definitions of key terms**

Terms are ordered as they appear in the main text and mostly defined after Margules & Pressey (2000) and Kukkala & Moilanen (2013).

- **Biodiversity trend indicator**: Statistic intended to provide information about trends in the state of biodiversity in a way that is easily understood. Common examples include areas of a remaining habitat, numbers of threatened and endangered species (e.g., IUCN Red List Index), and the UK wild bird index (based on population trends in wild breeding birds).

- **Conservation prioritization** (also known as spatial conservation prioritization)

- **Vulnerability**: Likelihood or imminence of the loss of a biodiversity feature.

- **Cost-effectiveness**: Difference made by actions divided by cost of delivering them.

- **Scarcity**: Degree to which biodiversity features (e.g., species or habitat types) are intrinsically scare and/or reduced relative to a reference state.

- **Complementarity**: Novelty or nonoverlap of biodiversity features added to an existing set.

- **Irreplaceability**: The marginal contribution of a biodiversity feature or action to a conservation goal. Irreplaceability is calculated stepwise (“iteratively”) because it depends on context, especially scarcity and complementarity.

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A similar determination of trend in the absence of intervention is needed to report the effectiveness of conservation actions. In both reporting and prioritization, estimating “business-as-usual” requires: (1) data measuring changes at a site without policy or management (e.g., Laurance *et al.* 2012) and/or (2) a prediction of the expected change in the absence of intervention (e.g., Nicholson *et al.* 2012). Many methods are available for predicting ecological changes and expressing their uncertainty (Sutherland 2006). One approach is to compare changes between matched areas with and without interventions (e.g., Joppa & Pfaff 2011).

**Assessing contributions to the state of biodiversity**

The difference made to the state of biodiversity as a consequence of pressures and conservation interventions depends on context. *Scarcity* is one aspect of context that matters because changes in rare or diminished features have greater impacts on the state of biodiversity. This is captured in the idea of “diminishing returns”, which are inherent in biological systems and reflect the fact that equivalent increases in an indicator will convey relatively greater changes in a biodiversity feature where the indicator starts at a smaller value (i.e., logarithmic-shaped curve). For example, population viability rises steeply with initial increases in numbers of individuals, and more slowly thereafter (Pimm *et al.* 1988). Therefore, each individual added to a population has a less positive effect on population viability than the one before.

The same logic dictates that the loss of the same area of habitat will eradicate more species in situations where fewer habitat remains than where more habitat remains. The contribution of changes to biodiversity goals is also influenced by redundancy among biological features and functions, or conversely *complementarity* (Pressey *et al.* 1993). For example, there are generally diminishing gains in the number of unique species protected when adding habitat to a protected area network, because common species are shared between areas. Similarly, the number of species lost in the clearance of two different habitat areas depends on the degree to which their species composition overlaps.

Trend indicators rarely account for context, making it difficult to gauge how much the state of biodiversity has truly changed. As a simple example, area of habitat lost is an indicator that is frequently reported with no other context. However, an amount of habitat lost in hectares says nothing about how much the habitat clearance compromised biodiversity goals, which depend on the pre-existing scarcity of habitats and the overlap in species composition among cleared and uncleared fragments (Figure 1).

In conservation prioritization, context-dependent contributions to conservation goals are calculated iteratively and termed *irreplaceability* (Pressey & Nicholls 1989; Ferrier *et al.* 2000). Although rarely applied in biodiversity reporting, the same approach can be used to combine disaggregated biodiversity data into indicators by accounting for the context-dependent contributions of change in individual features toward higher-level goals.
Figure 1 Conservation reporting relates indicators I to multiple features of biodiversity F, which ultimately sum together to comprise the state of biodiversity. (a) When context is ignored, it is assumed that equivalent changes in F (shown by arrows C₁ and C₂) arise from changes in I of equal magnitude (ΔI₁). (b) Scarcity influences the interpretation of indicators that are nonlinearly related to features of biodiversity. For example, species-area relationships predict an accelerating rate of species loss as I decreases. Thus, C₃ > C₄ when the context of habitat scarcity is considered.

For example, Overton et al. (2015) recently showed how protecting different combinations of sites would differentially promote the retention of a set of species, given the known locations, scarcity, and vulnerability of species’ populations across a broader region. Such an approach to reporting on the difference made to biodiversity therefore requires the same contextual data and a similar iterative calculation approach as conservation prioritization. This approach also ensures that indicators of the state of biodiversity capture those features actually prioritized by management and reveal the difference made by management relative to potential alternative sets of actions (including “business-as-usual”).

Accounting for cost-effectiveness

Conservation policies and management efforts are costly and usually constrained by resources. Furthermore, the cost-effectiveness (difference made-per-dollar; Box 1) of different conservation actions is extremely uneven; potentially varying by up to six orders of magnitude (Stephens et al. 2002; Balmford et al. 2005; Laycock et al. 2009; Gjertsen et al. 2014). In the conservation prioritization field, consideration of cost is recognized as essential for maximizing conservation achievement (e.g., Naidoo et al. 2006), and cost-effectiveness of different actions in different contexts can be forecast and compared in the same way as simple effectiveness, i.e., “difference made” (Stephens et al. 2002).

Incorporating cost-effectiveness in biodiversity reporting is also desirable. For example, transparency about the magnitude of cost associated with the difference made to biodiversity should facilitate greater accountability for resource allocation as well as policy and management adaptation. But costs are rarely included in biodiversity reporting, and, where present, total cash expenditure is usually all that appears (e.g., $2,703,000 expended on pest management in New Zealand [Department of Conservation 2012, p. 25]; $158,761,000 spent on conservation actions [Parks Canada 2012, p. 32]). Incorporating estimates of difference made and contextual contributions to higher-level goals into reporting, as we have suggested here, would also enable the cost-effectiveness of different conservation actions to be compared. Ideally, reporting this way would encourage greater attention to the selection of activities that deliver the greatest value for conservation investment.

Data for reciprocal reporting, prioritization, and management

The above discussion foreshadows that the data needed for reporting to guide conservation management and policy are broader than those usually collected for reporting on trend indicators alone, and generally align with three main needs.

First, predicting vulnerability under a business-as-usual scenario is a core requirement if reporting is to progress from portraying observed changes in features to showing how policy and management interventions make a difference. This requires empirical or modeled data on the distributions and abundances of pressures such as invasive pests and habitat loss, and data describing how changes in pressures influence biodiversity features. The blend of long-term management and (often academic) experiments needed to generate these data may be best achieved through partnerships between basic research and applied management organizations (Krebs 1991).

Establishing the contexts relevant for reporting on the state of biodiversity also requires data on the scarcity and complementarity of biodiversity features. Specific needs
include data describing past and present distributions and abundances of features across landscapes, such as species and habitat types. Principles of conservation prioritization suggest that data collection will be most efficient if focused on features that are now scarce and little-shared among sites, because these will be more irreplaceable and make a greater difference to the overall state of biodiversity. However, as sparse and rare components of biodiversity are often challenging to measure, improving the effectiveness of sampling techniques is an associated research challenge.

Finally, reporting the cost-effectiveness of interventions undertaken during a reporting period would require data on the costs of those actions. A particular challenge for practitioners is that costs of conservation actions often extend beyond simple government or agency expenditure into the realm of opportunity costs, and represent a type of information rarely considered in biodiversity reporting.

Although collecting these three types of data presents challenges, we note that all are core requirements for sound conservation prioritization. Thus, biodiversity reporting and conservation prioritization may be seen as reciprocal activities that are practically linked by shared requirements for information gathering. This conceptual and practical connection has not been explicitly recognized until now.

**Conclusions**

How effectively biodiversity reporting influences conservation outcomes is likely to depend not only on how well it spurs public awareness, but also on its ability to inform adaptive management and promote accountability for outcomes. Trend indicator reporting may achieve public awareness but systematically fail to promote adaptive management and accountability. We have suggested that reporting could better guide conservation action if it incorporated approaches developed for conservation prioritization, listed three basic building blocks of a shared framework, and provided a provisional outline of the types of data required (Table 1). A framework that recognizes the reciprocity of reporting, prioritization, and management should improve biodiversity outcomes, but requires broader information and analyses than previously appreciated for reporting purposes. We suggest that the potential for conservation reporting to inform policy and management will not be realized without such a framework.

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**References**

Balmford, A., Bennun, L. & ten Brink, B. et al. (2005). The convention on biological diversity’s 2010 target. *Science, 307*, 212-213.

Butchart, S.H.M., Walpole, M. & Collen, B. et al. (2010). Global biodiversity: indicators of recent declines. *Science, 328*, 1164-1168.

Department of Conservation. (2012). *Annual report: for the year ended 30 June 2012*. New Zealand Government, Wellington.

Ferrier, S., Pressey, R.L. & Barrett, T.W. (2000). A new predictor of the irreplaceability of areas for achieving a conservation goal, its application to real-world planning, and a research agenda for further refinement. *Biol. Conserv.*, 93, 303-325.

Gardner, T. (2010). Monitoring forest biodiversity: improving conservation through ecologically responsible management. Routledge, London.

Gjertsen, H., Squires, D. & Dutton, P.H. et al. (2014). Cost-effectiveness of alternative conservation strategies with application to the Pacific leatherback turtle. *Conserv. Biol.*, 28, 140-149.

Grantham, H.S., Bode, M. & McDonald-Madden, E. et al. (2009). Effective conservation planning requires learning and adaptation. *Front Ecol. Environ.*, 8, 431-437.

Hammond, A., Adriaanse, A. & Rodenburg, E. et al. (1995). Environmental indicators: a systematic approach to measuring and reporting on environmental policy performance in the context of sustainable development. World Resources Institute, Washington, DC.

Jones, J.P.G., Collen, B. & Atkinson, G. et al. (2011). The why, what, and how of global biodiversity indicators beyond the 2010 target. *Conserv. Biol.*, 25, 450-457.

Joppa, L.N. & Pfaff, A. (2011). Global protected area impacts. *Proc. Roy. Soc. B*, 278, 1633-1638.

Krebs, C.J. (1991). The experimental paradigm and long-term population studies. *Ibis*, 133, 3-8.

Kukkala, A.S. & Moilanen, A. (2013). Core concepts of spatial prioritisation in systematic conservation planning. *Biol. Rev.*, 88, 443-464.

Laurance, W.F., Useche, D.C. & Rendeiro, J. et al. (2012). Averting biodiversity collapse in tropical forest protected areas. *Nature, 489*, 289-294.

Laycock, H., Moran, D. & Smart, J. et al. (2009). Evaluating the cost-effectiveness of conservation: The UK Biodiversity Action Plan. *Biol. Conserv.*, 142, 3120-3127.

Margules, C.R. & Pressey, R.L. (2000). Systematic conservation planning. *Nature, 405*, 243-253.

Moilanen, A., Wilson, K.A. & Possingham, H.P. (2009). Spatial conservation prioritization: quantitative methods and computational tools. Oxford University Press, Oxford.

Naidoo, R., Balmford, A. & Ferraro, P.J. et al. (2006). Integrating economic costs into conservation planning. *Trends Ecol. Evol.*, 21, 681-687.

Nichols, J.D. & Williams, B.K. (2006). Monitoring for conservation. *Trends Ecol. Evol.*, 21, 668-673.
Nicholson, E., Collen, B. & Barausse, A. et al. (2012). Making robust policy decisions using global biodiversity indicators. *PLoS ONE, 7*, e41128.

Overton, J.M., Walker, S. & Price, R. et al. (2015). Vital sites and actions: an integrated framework for prioritizing conservation actions and reporting achievement. *Diversity Distrib.*, 21, 654-664.

Parks Canada. (2012). *Parks Canada agency – performance report 2011–12*. Government of Canada, Gatineau.

Pereira, H.M., Ferrier, S. & Walters, M. et al. (2013). Essential biodiversity variables. *Science, 339*, 277-278.

Pimm, S.L., Jones, H.L. & Diamond, J. (1988). On the risk of extinction. *Am. Nat.*, 132, 757-785.

Platt, J.R. (1964). Strong inference. *Science, 146*, 347-353.

Pressey, R.L. & Nicholls, A.O. (1989). Efficiency in conservation evaluation: scoring versus iterative approaches. *Biol. Conserv.*, 50, 199-218.

Pressey, R.L., Humphries, C.J. & Margules, C.R. et al. (1993). Beyond opportunism: key principles for systematic reserve selection. *Trends Ecol. Evol.*, 8, 124-128.

Stephens, R.T.T., Brown, D.J. & Thornley, N.J. (2002). Measuring conservation achievement: concepts and their application over the Twizel area. *Sci. Conserv.*, 200, 1-114.

Sutherland, W.J. (2006). Predicting the ecological consequences of environmental change: a review of the methods. *J. Appl. Ecol.*, 43, 599-616.

Walpole, M., Almond, R.E.A. & Besançon, C. et al. (2009). Tracking progress toward the 2010 biodiversity target and beyond. *Science, 325*, 1503-1504.