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From displacement activities to evidence-informed decisions in conservation

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ABSTRACT

This paper highlights a disjunction between the basic motivation of conservation planners, policy-makers, and managers, which is to make a positive difference for biodiversity, and many of our day-to-day activities, which are tangential (at best) to the goal of avoiding biodiversity loss. At the core of this problem is the use of conservation measures (inputs, outputs, and outcomes) that do not explicitly address conservation impact, and thus risk undermining its achievement. These measures are used to formulate policy targets and operational objectives, gauge progress towards them, and identify priorities for action. In particular, the pervasive use of representation of biodiversity features as a sole basis for identifying priorities, and the considerable global effort directed towards increasing protected-area extent and assessing protected-area management effectiveness, exemplify that much conservation decision-making is founded more on belief systems than evidence. Measures such as the extent or representativeness of protected areas risk misdirecting conservation actions towards areas of low impact and misleading decision-makers and the public about conservation progress. To promote more effective, evidence-informed decision-making, analytical evidence can and should be used to test and refine decision-makers' implicit models of the world, focusing on predicting conservation impact - the future difference made by our future actions - to increase our effectiveness and accountability.

1. Introduction

When frustrated, thwarted, or faced with conflict, animals exhibit behaviour - termed displacement activity - out of context with, and apparently irrelevant to, their prevailing situations (Delius, 1967). Displacement activities by birds under threat or in conflict include feeding and nest-building movements, preening, and sleep, interpreted as outlets through which frustrated drives can be expressed (Tinbergen, 1952). Displacement activities have been described in response to stress in non-human primates (Maestripieri et al., 1992) and humans (Mohiyeddini and Semple, 2013). Could it be that conservation professionals exhibit collective forms of displacement activity? Have we adopted irrelevant responses to the irreconcilable tension between needing to save biodiversity, and the difficulty in doing so in the face of the combined erosive force of human numbers, extractive activities, invasive species, and climate change? Are we retreating to activities that are immediately attainable, personally profitable, and politically advantageous at the expense of helping biodiversity to persist?

Whitten et al. (2001) asked a similar question of conservation biologists. They also posed a more specific and confronting question "… if conservation biology is ineffective in helping to stop something as globally significant as the devastation of Indonesian forests, then what, please, is the point of it?" This might seem a harsh criterion by which to judge a scientific field but, in the end, conservation science will be judged by how much difference it has made, not by the shorter-term criteria of publications, conference presentations, research grants, and personal advancement. Conservation policy will be judged in the same way, not by the achievement of protected-area targets unrelated to making a difference (Pressey et al., 2015). The same is true of protected-area management, currently assessed by agreed criteria (Leverington et al., 2010) that appear unrelated to saving biodiversity (Coad et al., 2015). These three areas of conservation endeavour are analogous to medical research, policy, and practice, which would be judged harshly if they failed to reduce human suffering and death.

This paper examines whether conservation policy, planning, and management are making a positive difference for biodiversity, or whether they constitute displacement activities in the face of biodiversity loss. Section 2 defines types of "measures". We use this term to refer to policy targets, such as those for protected areas under the Convention on Biological Diversity (Convention on Biological Diversity, 2010), quantitative objectives for operational decisions in protected-area management and identification of priorities through conservation planning (Pressey and Bottrill, 2009), and the application of targets and objectives to gauge progress in conservation. Commonly used measures are extent or representativeness of protected areas (Pressey et al., 2015). Section 3 critically reviews the types of measures that dominate decision-making in conservation, using the lens of conservation impact, which is the difference made by conservation actions (Ferraro and Pattanayak, 2006). Section 4 revisits the notion of displacement activities, concluding that, at least analogously, they characterise much of our decision-making, which is founded more on belief systems than evidence. Sections 3 and 4 highlight a disjunction between, on one hand, the basic motivation of policy-makers and conservation planners and managers and, on the other hand, many of our day-to-day activities. It seems reasonable to say that people working in conservation have set out to

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make a positive difference, but this motivation is not expressed in much of our work. As a consequence, we oversee avoidable loss of biodiversity. Section 5 proposes levels of evidence to replace belief systems in conservation and maps a way towards policy, planning, and management that directly address the fundamental goal of conservation impact. The broad goal of the paper is to contribute to discussion about how conservation decision-making can be more effective in minimising loss of biodiversity. The focus is on decisions about spatial management through formal protection and application of conservation actions within and outside protected areas (hereafter "conservation areas").

2. Types of conservation measures

This section defines four broad types of measures – inputs, outputs, outcomes, and impacts – with outcomes separated into three sub-categories (Fig. 1A). The definitions follow established terms in performance management (DAC, 2002; Margoluis et al., 2013) and impact evaluation (Ferraro, 2009). Placing measures into categories has two advantages. First, it groups measures that use data in similar ways to formulate targets and objectives, gauge progress, and set conservation priorities. Second, it helps to understand the roles of different measures in decision-making and their functional relationships to one another (Margoluis et al., 2013; Pressey et al., 2015).

Inputs are the resources invested in conservation programs, usually in the form of staff, time, and money. Outputs are the concrete, countable products of conservation actions. Examples are numbers or total km^2 of protected areas, numbers of boats available for patrols, km of fencing, or numbers of pest animals culled. At the operational levels of conservation programs and management of protected areas, outputs are things that can be safely promised in return for funding. Outcomes are the observed or assumed effects of conservation outputs. The most immediate and easily measured outcomes are those related to representation (or sampling) of species, ecosystems, or other elements of biodiversity (hereafter "features"). Outcomes in terms of levels of threats to biodiversity are meant to indicate the effectiveness of actions in separating biodiversity features from processes that jeopardise their persistence; this separation is implied, but not guaranteed, by representation in conservation areas. Outcomes for the state of biodiversity convey information of more direct interest than the previous measures: they can reflect the responses of features to actions, which are not always proportional to threat reduction (Tulloch et al., 2015). Outcomes for threats and biodiversity are typically measured only within conservation areas or systems of conservation areas, at a single point in time or as trends over time (Pressey et al., 2015). The "impacts" of Margoluis et al. (2013) are categorised here as outcomes for biodiversity because they are not necessarily based on a comparison between conditions inside conservation areas and those outside.

Impacts, as defined here, are the "value added" of conservation: the effects of actions on one or more intended (or unintended) outcomes, over and above the counterfactual (Ferraro, 2009; Maron et al., 2013) of no action or a different action (Ferraro and Pattanayak, 2006). Impacts are therefore measures of difference (Fig. 1B) expressed, for example, as percentages of protected-area systems that avoid loss of forest cover (Andam et al., 2008) or the amount of potential loss of biodiversity in a region that was avoided by actions (Pressey et al., 2015). This definition brings conservation into line, as proposed by Ferraro and Pattanayak (2006), with very extensive applied research on impact evaluation in development aid, medicine, and education (Banerjee and Duflo, 2009; White, 2009). Importantly, this definition of impacts contrasts with that in the results chains of performance management (Margoluis et al., 2013) in which "impacts" are eventual outcomes for biodiversity (Fig. 1A).

Of the methods used to estimate impacts (Ferraro and Hanauer, 2014), perhaps the most intuitive is matching. Matching involves choosing sites within conservation areas and matching each to a site outside, taking care that the inside-outside pairs are very similar in characteristics (e.g. slope, distance from markets, extent of unaltered habitats, inherent suitability for selected species) that could affect conditions of interest (e.g. forest cover, abundance of vulnerable species). The conditions of the outside sites are then estimates of the expected conditions of their matched inside sites had conservation actions not been taken (e.g. Andam et al., 2008).

Only impacts allow decision-makers to understand how much difference they have made or could make. The key distinction between impacts (Fig. 1B) and outcomes (Fig. 1A) is the estimation of impacts by comparing conditions within conservation areas to those expected without conservation actions (Pressey et al., 2015). Most impact evaluations have been retrospective, providing lessons for the future; but planners and managers must also move towards predicting impacts – essentially predicting conditions across regions with and without conservation actions – to identify priorities for action that reflect the potential to avoid future loss of biodiversity.

The reliability of impact estimates depends on how rigorously counterfactual conditions are identified. Comparisons between protected sites and those just outside protected-area boundaries (Bruner et al., 2001), for example, can be unreliable for several reasons. First, sites just outside boundaries can have much higher probabilities of losing biodiversity than those inside if boundaries follow, as they often do, discontinuities such as breaks in slope, changes in soil type, or edges of reefs. The resulting estimates of impact can be substantially inflated (Andam et al., 2008; Geldmann et al., 2013). Second, across-boundary comparisons are affected by localised interaction effects, either through protection supplementing biodiversity outside (Harrison et al., 2012) or displacing extractive activities from within conservation areas to areas outside (Bode et al., 2015). Counterfactual estimates can also be simply misconceived. The measure of "true" conservation progress (McDonald-Madden et al., 2009) is based on a ratio of conservation to loss of features over a defined period, exemplifying what Game et al. (2014) described as a good solution to the wrong problem. The measure fails to convey information about impacts because it does not estimate how much loss would have occurred in the absence of conservation actions and how much of that loss was avoided.

Even rigorous assessments of impacts, however, come with assumptions and limitations. For example, some protected sites might have no good outside matches (Pfaff et al., 2009), and there is a tradeoff between quality of matches and number of matched sites (Ahmadia et al., 2015). Avoided deforestation might under- or overestimate impact related to variables not detectable from remote sensing, such as density of understorey important to some animal species (Vincent, 2016), although this problem is avoided by evaluations based on field surveys (Gaston et al., 2008; Geldmann et al., 2013). While the effects of protection on local displacement of threats (also referred to as "spillover" or "leakage") are understood (Ewers and Rodrigues, 2008) and can be accounted for in counterfactual estimates (Andam et al., 2008), some displacement could extend well beyond study regions, even internationally (Henders and Ostwald, 2012). Further, identifying potential mechanisms for observed impacts and reaching conclusions about causality is not always straightforward (Ferraro and Hanauer, 2014). Notwithstanding these limitations, the considerable advantage of impacts over other measures is that they reflect the basic purpose of protected areas. Like any emerging area of research and development, evaluation of conservation impacts will become progressively refined, and more quickly if impacts become a focus of science and policy.

3. What do types of measures tell us about conservation impact?

The five types of measures in the results chain (Fig. 1A) dominate decision-making in conservation. This section reviews them in terms of the information they provide about impact and the risks of using them for conservation decisions.

3.1. Questions to ask about types of measures

If a measure is used in policy, planning, or management to set targets and objectives, gauge progress towards them, and identify priorities, it should be scrutinised carefully beforehand. There is, after all, a great deal at stake. If measures misdirect limited conservation resources, those resources will be wasted, and evident failure could compromise future conservation efforts. Mistakes also mean that biodiversity will be lost unnecessarily because protection has not been afforded to the species and ecosystems most in need. Often, mistakes cannot be fixed later. In many cases, the consequent loss of biodiversity is rapid and irretrievable. In other cases, longer-term attrition in areas that should have received conservation attention might be arrested only with additional actions that require more funding than is available or more conservation areas than are socially and politically acceptable.

Two questions should be asked before using measures in conservation:

1. Could the measures misdirect conservation actions?
2. Could the measures overstate conservation progress?

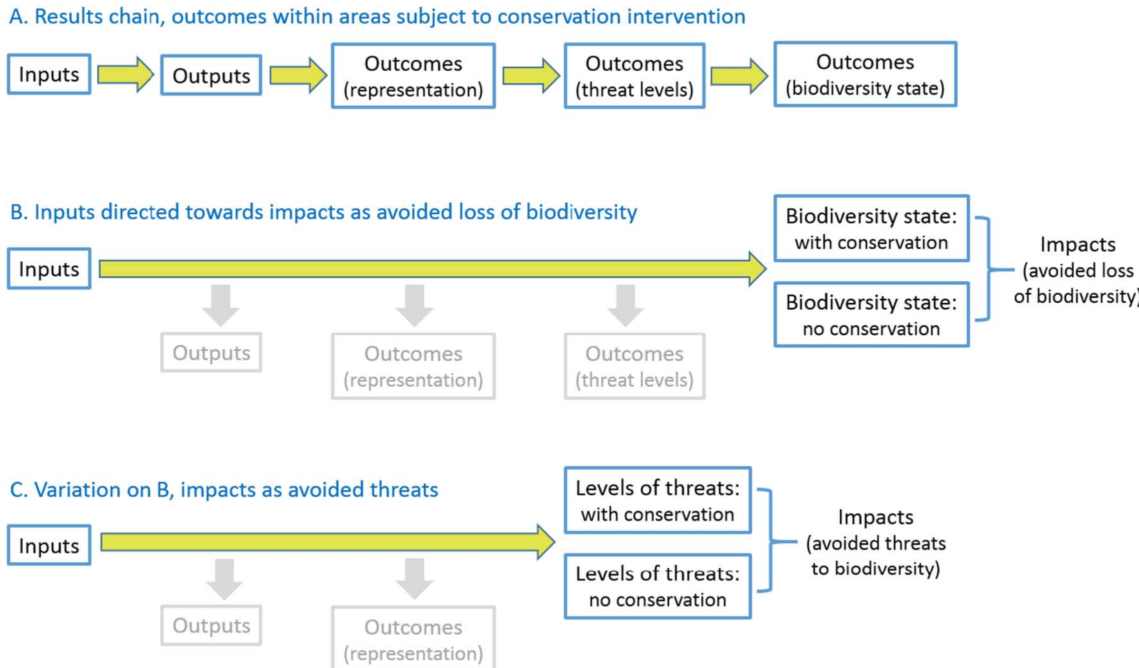

Fig. 1. Relationships between types of measures. A is a results chain (Margoluis et al., 2013) that does not include impacts as defined here. B directs inputs towards impacts measured as avoided loss of biodiversity, with outputs and outcomes achieved incidentally. When data on biodiversity are too sparse or generalised, it might be necessary to achieve impact by directing inputs towards avoiding threats (C). Goals, actions, assumptions, and feedbacks (Pressey et al., 2015) have been omitted for simplicity.

In one sense, there is redundancy here. If a measure does not carry enough information about impact, it could be used, unwittingly or deliberately, both to misdirect actions and wrongly portray progress. Nonetheless, asking both questions can be useful to fully appreciate the risks of using measures to influence conservation decisions (Table 1). The risks are related not just to inaccuracies about impact, which would imply both over- and under-estimates. Specifically, if a measure is uninformative about impact, the main risks are misdirecting conservation actions to areas of low impact (under-achievement) and exaggerating conservation progress (overstatement) (Pressey et al., 2015).

The assumption in framing risks in this way is that, unless constrained by measures that are informative about impact, many conservation actions, and especially protected areas, will gravitate to places where impact is small. This assumption is well supported. The residual nature of protected areas – meaning their concentration in areas least suitable for conversion or extraction of natural resources – is a safe generalisation on land and in the sea (Devillers et al., 2015; Joppa and Pfaff, 2009). Factors that drive protection to the margins of human activities and requirements include socioeconomic interests that override conservation concerns, political expediency, and a focus by some conservation groups on extensive, remote, wild places (Armesto et al., 1998; Barnard et al., 1998; Pressey et al., 2000; Virkkala et al., 1994). The evidence is strong that residual "protection" of places least in need of conservation actions can lead to small impacts. For example, Andam et al. (2008) estimated that some 91% of the forest in Costa Rica's protected-area system would have been retained over 37 years in the absence of protection, so the difference made was just 9% of the total protected-area extent. Similarly, in Brazil's Acre State, Pfaff et al. (2014) estimated the difference made to deforestation at 1% or less of the protected-area system over five years. While protection tends towards residual areas, the biodiversity in areas most urgently in need of protection is being lost, much of it irretrievably.

3.2. How do measures measure up?

To complement the assessments here, Table 1 answers the two questions posed in Section 3.1 about the types of measures in Fig. 1A. This section focuses particularly on outputs and representation outcomes because of their large influence on policy targets and planning objectives, respectively.

3.2.1. Inputs

Intuitively, more inputs should lead to better results for conservation. The ultimate value of inputs, however, depends on where, when, and how they are directed. Inputs directed at any of the other measures in Fig. 1A could fail to generate impacts because of the limitations of those other measures (below). There is no guarantee of positive and substantial impacts unless impacts are themselves specified as objectives and actions are taken accordingly (Fig. 1B). The links between inputs and impacts have rarely been demonstrated (Geldmann et al., 2013). Notably, inputs feature prominently in assessments of management effectiveness of protected areas (Leverington et al., 2010, including their "process" variables), which might help to explain why few studies have demonstrated a relationship between management effectiveness and impact. Coad et al. (2015) identified only three published studies, all terrestrial, that compared management effectiveness to impacts of protected areas measured with counterfactual designs, and none of those studies found a relationship. Evidence is emerging, however, for a link between management resources and ecological impact of marine protected areas (Gill et al., 2017), albeit across a small sample.

3.2.2. Outputs

A belief in outputs implies a belief that quantity equals quality in conservation, which is intuitively and demonstrably wrong. Intuitively, numbers of feral animals culled or hectares of weeds removed are relevant only if reductions are sufficiently large and sustained to achieve an adequate positive response by affected native species. The value of fences built to exclude domestic stock or feral animals depends less on their length and more on where and why they were placed, and what difference they make to the overall persistence of susceptible species. Achieving quantitative policy targets for protected-area extent comes with the risk of failing to achieve other important benefits that are presently framed vaguely and qualitatively (Watson et al., 2015).

Quantitatively, the value of marine protected areas depends not on their extent but on how much difference they make to activities that negatively affect biodiversity (Devillers et al., 2015). Very small percentages of the total extents of terrestrial protected areas contribute to impact (Pressey et al., 2015). Increases in number and extent of protected areas through time can be accompanied by poor or even worsening progress in conservation of threatened ecosystems (Pressey and Taffs, 2001; Pressey et al., 2002). Across regions, larger percentages under formal protection have been correlated with stronger biases away from landscapes most in need of protection (Pressey et al., 2000).

Given these realities, it is important to ask why, for example, one of the most influential protected-area targets globally (Target 11, Convention on Biological Diversity, 2010), echoed in numerous regional and national

Table 1
Summary assessment of types of measures in relation to informativeness about conservation impact.

Measures	Could they misdirect conservation actions?	Could they overstate conservation progress?
Inputs (e.g. financial commitments, staff, infrastructure)	**Yes**. Explicit direction of inputs to impacts is rare. More commonly, conservation resources are committed to measures in the results chain (Fig. 1A), and especially to outputs. The risk of inputs being misdirected is therefore considerable. There is no evidence that management effectiveness of protected areas, which is dominated by considerations of inputs, is related to impact on land (Coad et al., 2015), although some evidence is emerging for marine protected areas (Gill et al., 2017).	**Yes**. Reporting on investments in conservation is meaningful only if it can be demonstrated that they have been or will be directed to achieving impacts. Demonstration of this link is rare. Apparently large investments in conservation can impress undiscerning audiences while falling well short of actual requirements, even for representation outcomes (Adams et al., 2011).
Outputs (e.g. extent of protected areas)	**Yes**. There is no necessary relationship between the quantity of conservation actions and their impacts (Pressey et al., 2015), and some studies indicate a negative relationship (Pressey et al., 2000; Pressey et al., 2002). Directing conservation resources towards outputs, as encouraged by many donors and funding programs, could have perverse results by encouraging low-impact actions (Visconti et al., 2015).	**Yes**. Reporting requirements for many funding programs mistake outputs for measures of conservation progress. Outputs, most notably in the form of extent of protected areas, are routinely used to overstate conservation progress by obscuring residual protection. Rigorous evaluations indicate that impact is generally a small percentage of protected-area systems (Pressey et al., 2015).
Outcomes for representation (e.g. number of ecosystems or species sampled in protected areas)	**Yes**. Measures of representation outcomes are counts of features and their relative coverage by conservation actions. Aiming to increase the number of represented features risks protecting those that are easiest to protect at the expense of those most urgently in need of protection, with urgency indicated by extensive past reductions and/or rapid ongoing declines (Fig. 2). Sets of proposed conservation areas that represent all features are rarely implemented completely. Partial implementation is likely to focus on least contentious areas that contribute least to impact.	**Yes**. Increasing values for representation outcomes can give the false impression of conservation progress in two ways: by counting features regardless of the urgency of their protection; and by counting features that have been given minimal and inadequate representation in conservation areas. Representation of features in conservation areas can increase over time or across regions in parallel with unchanged or increasing conservation biases away from features most in need of protection (Pressey et al., 2000; Pressey and Taffs, 2001; Pressey et al., 2002).
Outcomes for levels of threat (e.g. reduced poaching of animals in protected areas, reduced deforestation, reduced extent of invasive animals)	**Yes**. Priorities for reduction of threats might not lead to positive responses of biodiversity. More fundamentally, priorities to reduce threats (conservation impact) require explicit analyses across whole planning regions, including: spatially explicit prediction of levels of threats in the absence of actions; and choices of actions, places, and times to maximise avoided threats relative to cost. Without these analyses, actions could be misdirected (Pressey et al., 2015) by focusing on: areas where threats are most easily mitigated, regardless of the difference made; areas subject to severe threats that cannot feasibly be mitigated; or areas where impact can be achieved locally but at lower levels than areas not assessed.	**Yes**. Reported reductions in threats might not reflect positive responses of species or ecosystems of conservation concern. More fundamentally, state of or trends in threats cannot be attributed to conservation actions without rigorous impact evaluation that compares areas subject to actions with those lacking actions, after accounting for confounding influences (Pressey et al., 2015). The key question is: to what extent would the observed state or trends have arisen, from factors internal or external to the conservation area, without the conservation actions?
Outcomes for state of biodiversity (e.g. numbers of fish in marine protected areas, increases in numbers of arboreal mammals in areas subject to actions)	**Yes**. Priorities for avoided loss of biodiversity (conservation impact) require the same explicit analyses as above, with the addition of predicting responses of species and/or ecosystems to reductions in threat levels. Without these analyses, belief systems about "important" areas for biodiversity conservation remain untested (Pressey et al., 2015). Even the apparent logic of prioritisations based on explicit criteria (e.g. Key Biodiversity Areas, Foster et al., 2012) and quantitative predictions (e.g. irreplaceability and vulnerability, Margules and Pressey, 2000) might constitute no more than preconceptions and biases without testing to demonstrate that the recommended approach will avoid more loss than alternative approaches.	**Yes**. State of or trends in biodiversity cannot be attributed to conservation actions without rigorous impact evaluation that compares areas subject to actions with those lacking actions, after accounting for confounding influences (Pressey et al., 2015). The key question is: to what extent would the observed state or trends have arisen, from factors internal or external to the conservation area, without the conservation actions?

targets, is framed as extent. Equally, we should ask why IUCN's previous Director-General stated, in the lead-up to the sixth World Parks Congress, that a doubling in number of protected areas since the previous Congress was good news (Marton-Lefèvre, 2014), or why major funding programs for environmental management report on inputs, outputs, and activities, but not on impacts (Park et al., 2013).

The attractiveness of outputs as measures, with their simplicity and certainty, risks diverting attention from the real work of conservation. When policy targets or operational objectives are set for outputs in their own right, the actual needs of conservation can be forgotten (Melick et al., 2012). The real test is whether the actions motivated by a measure make a difference relative to the counterfactual of no actions (Fig. 1B). In this context, outputs are incidental: necessary but not sufficient, and not quantities to be maximised for their own sake (Pressey et al., 2015). A continuing focus on outputs could further compromise conservation efforts. Time-bound targets for protected-area extent (Convention on Biological Diversity, 2010), for example, come with the substantial risk of exacerbating the trend towards residual protection by focusing attention on parts of the land and sea that are easiest and quickest to protect, and away from those most in need of protection (Barnes, 2015; Watson et al., 2015). Recent modelling indicates that achieving the Aichi target of 17% protected-area coverage, in the absence of species-specific objectives for avoided loss, could have negligible or even negative conservation impacts for threatened species (Visconti et al., 2015). This result arose from focusing new protected areas on places with low cost (indicated by low human densities), thereby displacing expanding agriculture and urban development to places with high human densities that were nonetheless rich in threatened species.

3.2.3. Representation outcomes

Measures of representation, despite their influence on policy (CTI-CFF, 2013; Driver et al., 2005; National Reserve System Task Group, 2009) and core role in shaping the burgeoning field of systematic conservation planning (Margules and Pressey, 2000), can be misleading about past and future impact. These measures include numbers of features with conservation objectives achieved (Pressey et al., 2002), equality of representation across features (Barr et al., 2011), and depictions of biodiversity in multivariate environmental space (Faith and Walker, 1996). The discussion below focuses on representation of biodiversity pattern (e.g. maps of ecosystems or species occurrences), but applies equally to locating and configuring areas for the persistence of biodiversity processes and ecosystem services.

Representation outcomes are essentially counts of features (or metrics of coverage of environmental space) and their relative levels of protection. Framing targets and objectives on representation outcomes therefore ignores the relative urgency of protection of features, assuming implicitly that all will eventually be adequately represented. This assumption is unreliable. Full representation can be posed as a challenge for governments (Taylor et al., 2014) and pursued through objectives for conservation planning (Tear et al., 2005), but its achievement in the real world is rare. Many plans are simply not followed by implementation (Knight et al., 2006a). Even when they are, strong forces resist adequate conservation action over landscapes and seascapes with commercial potential. Application of representation to forest conservation in New South Wales, even with political support, fell short of objectives for forest ecosystems of most value to industry (Pressey et al., 2002). The large expansion of marine protected areas in Australian waters in 2012, ostensibly directed at representation, placed no-take zones where they would least interfere with commercial extraction of resources (Devillers et al., 2015). Even the exemplary re-zoning of the Great Barrier Reef Marine Park, while achieving at least 20% representation of each marine bioregion in no-take zones, placed those zones on the parts of soft-bottom bioregions with least value for trawling, leaving questions about true representativeness unanswered (Devillers et al., 2015).

Such real-world constraints mean that, while some features are represented increasingly in conservation areas over time, those more vulnerable to extraction and outright conversion tend to remain unprotected, at risk of further decline, and more likely to have conservation objectives compromised before they can be achieved (Fuller et al., 2007; Visconti et al., 2015). More important than the number of features protected at any time, therefore, is their identity: are they the ones most urgently needing

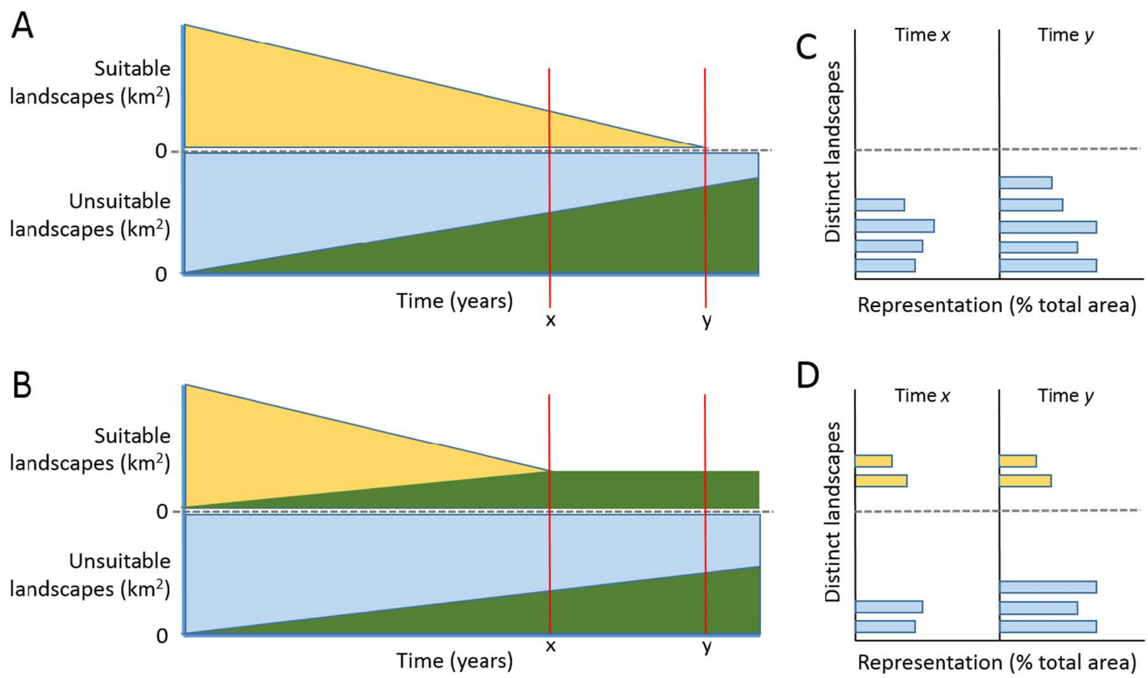

Fig. 2. Two hypothetical time-series of increasing representation of landscapes in protected areas. Orange indicates total (A,B) or protected (C,D) extent of landscapes suitable for agricultural development. Blue indicates the same for landscapes unsuitable for agriculture. Green in A and B indicates extent of protection of landscapes in the two categories.

protection, or the ones that were easiest to protect? Consider two strategies for scheduling the conservation of landscapes in a region (Fig. 2). The first strategy (Fig. 2A) gives priority to protection of landscapes unsuitable for agriculture. Over time, large portions of these landscapes are protected so, at time *y*, no options remain for protection of suitable landscapes, all of which have been converted to agriculture. The second strategy (Fig. 2B) affords early protection to some landscapes suitable for agriculture so, at time *x*, further conversion to agriculture is preempted. At both times *x* and *y*, the representativeness of the first strategy (Fig. 2C), expressed as the overall number of landscapes under protection and their percentages covered, is equal to or better than that of the second strategy (Fig. 2D), but only the second strategy achieves impact. Of course, real landscapes are not binary in terms of suitability for conversion and need for protection, but this simple example reflects salient aspects of reality: conversion of landscapes to human uses is highly selective (Pressey et al., 2000); protection is strongly biased towards unsuitable landscapes (Joppa and Pfaff, 2009); and time-series studies show that representation can increase while protection bias towards unsuitable landscapes remains stable or increases (Pressey and Taffs, 2001; Pressey et al., 2002). The upshot is that progress towards targets and objectives framed as representation outcomes is likely to over-estimate conservation impacts.

For similar reasons, conservation priorities for representation outcomes can be unreliable guides for conservation actions. Typically, these priorities are designs for areas that collectively achieve, at least virtually in planning software, representation objectives for all features of concern (Watts et al., 2009). Implementation, however, is another matter. Even when implementation is rapid, it is seldom complete (Devillers et al., 2015; Pressey et al., 2002). More typically, conservation designs are implemented incrementally over years or decades (Pressey et al., 2013), during which some features continue to be lost, more or less predictably. In this context, political pragmatism and the need for organisations to demonstrate early progress pose the substantial risk that more residual, and less costly and contested, parts of designs will be earmarked for early, and perhaps sole, attention. This approach could be depicted as progress by counting represented features, but it compromises impact by failing to intervene in avoidable loss.

Perverse consequences are also likely if conservation priorities are based on other measures of representation outcomes. Protection equality can be measured in many contexts, including equality of ecoregion protection within countries (Barr et al., 2011) or equality of mapped habitats within ecoregions. At such coarse resolutions, high equality, implying low priority, could disguise a few unprotected threatened features in need of urgent conservation actions; and low equality, implying high priority, could be improved by residual protection. Modelled tradeoffs between representation of features and cost (Ando et al., 1998; Faith and Walker, 1996; Gurney et al., 2015) invite residual outcomes by focusing conservation attention on the cheapest features, likely to be concentrated in areas with least exposure to imminent threats from conversion or extraction (Newburn et al., 2006).

3.2.4. Outcomes for threat levels

Outcomes related to threats add information to representation outcomes (Fig. 1A), reflecting the role of conservation actions, not just to intersect with examples of features, but to separate them from processes that jeopardise their persistence (Margoluis et al., 2013). Examples are threat levels following establishment of new protected areas, management to mitigate external pressures on established protected areas, or incentives to reduce fishing or logging (Salafsky and Margoluis, 1999). Examples of threat-related priorities, with limited information on biodiversity responses, include areas where actions would yield large reductions in deforestation (Salafsky and Margoluis, 1999) or removal of potentially harmful invasive species. Advantages of measuring outcomes for threats instead of those for state of biodiversity (Salafsky and Margoluis, 1999) can be offset by failure of the causal link between reducing threats and improving conditions for biodiversity (Pressey et al., 2015; Tulloch et al., 2015). More fundamentally, though, threat outcomes do not tell decision-makers about impacts (Pressey et al., 2015). Impacts are the differences in threat levels attributable to conservation actions, requiring a comparison between areas with the actions and those lacking them (Fig. 1C). Without that comparison, observed increases or reductions in threats in conservation areas could be due to factors, such as trends in markets or human populations, unrelated to local conservation actions. Without predicting differences that actions could make, threat-related impacts might not be achieved by conservation investments.

3.2.5. Outcomes for state of biodiversity

Outcomes related to the state of biodiversity add further information to those earlier in the results chain (Fig. 1A). Biodiversity state or trends are observed directly (Geldmann et al., 2013), not inferred from information on threats. The focus is on the extent, condition, abundance, or likely persistence (ideally estimated from population viability analysis, Frankham et al., 2014) of features subject to conservation actions, with data collected remotely (Nagendra et al., 2013) and/or from field surveys (Gaston et al., 2008). Priorities for biodiversity outcomes are meant to identify places where the biodiversity benefits of actions would be particularly high. A high-profile method for identifying priorities is to delineate Key Biodiversity Areas (Foster et al., 2012) – areas with globally significant populations of vulnerable species - now endorsed by the World Conservation Union (IUCN, 2015). Other proposed priorities for biodiversity are places that offer cost-effective gains in persistence (Polasky et al., 2008) and with intersections of high threat and biodiversity value (Margules and Pressey, 2000). These observed or expected outcomes, however, suffer from the same limitation as those for threats (Pressey et al., 2015): unless compared with counterfactual estimates (Fig. 1B) - retrospective or predicted - they are not informative about avoided loss (or the real effectiveness of past actions), and priorities could focus limited resources on the wrong places.

4. Policy, planning, and management as displacement activities?

Tinbergen (1952, p. 24) and Maestripieri et al. (1992, p. 969) considered the essence of displacement activities to be their irrelevance, or lack of relationship to normal functions or motivations. Decision-making in conservation differs from strictly defined displacement activity in two respects. First, human displacement activities are typically minor and can be unconscious (Mohiyeddini and Semple, 2013), whereas much decision-making in conservation is not simple but collective, complex, and orchestrated. Second, like some apparently out-of-context activities of non-human primates, further study can reveal clear functions, including attaining status and security (Maestripieri et al., 1992). To these motivations in conservation could be added “busy work” (Ammer, 1997), or the need for activity that appears superficially useful, perhaps in the face of uncertainty about how to be effective.

These differences mean that, if there is a relationship between conservation decision-making and displacement activity, it is analogy, not equivalence. If the analogy applies, it remains centred on irrelevance, as argued by Whitten et al. (2001), but extends beyond the focus of those authors on conservation biology to cover policy and protected-area management. Lack of relevance reflects the human tendency for “question substitution” (Kahneman, 2012, p.97): faced with a difficult question, people tend to answer an easier one that avoids the need for intractable analysis. The answer to the easier question might be appropriate to the difficult one, but might also be irrelevant or seriously misleading. In conservation, answers to the question “How do we save biodiversity?” all too readily become focused on how to achieve something smaller and more manageable, but potentially unrelated to the real question and perhaps counterproductive.

Conservation is analogous to displacement activity to the extent that, instead of doing what is necessary for conservation, we do what seems politically pragmatic and professionally rewarding (Whitten et al., 2001), or even what seems possible and reasonable, but at the risk of compromising our supposed goal of making a positive difference. A stock-take of measures describing most decision-making in conservation (Table 1) indicates the extent of this risk. None are informative about conservation impact, so progress could be made towards them while achieving impact only incidentally, if at all. Any of them could be used, deliberately or unwittingly, to overstate conservation progress. Displacement activity arises when measures in the results chain (Fig. 1A) are seen as ends in themselves, not as potential means to the end, which has always been conservation impact (Fig. 1B).

Three examples, reflecting pervasive and influential belief systems, illustrate the extent to which current decision-making fails to address conservation impacts and risks undermining them. First, assessment of management effectiveness of protected areas has substantial momentum globally, with a high profile in IUCN's World Commission on Protected Areas and thousands of assessments completed (Leverington et al., 2010). But very few studies have tested the link between management effectiveness and impact, and no relationship has been found for terrestrial protected areas (Coad et al., 2015).

Second, equating increase in protected-area extent with conservation progress defies considerable evidence to the contrary. Aichi Target 11 (Convention on Biological Diversity, 2010) has helped to institutionalise this belief, and has been embraced by governments, NGOs, and donors in pursuit of environmental respectability and statements of commitment to conservation that avoid the difficulties of actually making a difference. Measuring conservation progress in km^2 is equivalent to measuring progress in health care by the number of patients treated, even though most of the treated people were healthy (because they were cheaper to treat) and most of the people needing treatment went without.

Third, although representation has been fundamental to systematic conservation planning since the field's inception (Pressey, 2002), as long as we see representation as an end in itself rather than a possible means to achieve impact (Fig. 1B), it is a milestone that has become a millstone. A small amount of critical thinking reveals that representation is problematic as a measure of progress (Fig. 2) and as a sole basis for identifying priorities (Pressey et al., 2015). Diamond (1976) described the problem succinctly: “the question is not which refuge system contains more total species, but which contains more species that would be doomed to extinction in the absence of refuges.” Even with its limitations, representation has strong policy traction (Jenkins and Joppa, 2009; National Reserve System Task Group, 2009; Woodley et al., 2012). Despite recent refinements in the scope of decision-support tools to incorporate threats, economics, alternative actions, and biological persistence (Kareksela et al., 2013; Moilanen et al., 2011; Watts et al., 2009), representation remains at the core of considerable scientific effort to answer minor variations on artificially constructed questions in the virtual world of conservation-planning software, much of it of uncertain real-world relevance. One motivation for the disproportionate scientific emphasis on representation is the reward system for demonstrating technical prowess in publications, effectively devaluing critical understanding of real-world problems. Another motivation is the wide availability of free software, such as C-Plan (Pressey et al., 2009), Marxan (Ball et al., 2009), and Zonation (Moilanen, 2007), developed for representation. Maslow's (1969, p. 15) comment seems relevant: “… it is tempting, if the only tool you have is a hammer, to treat everything as if it were a nail.”

The barrier to basing decision-making on conservation impact is not lack of information. Residual conservation - the problem that impact evaluation reveals and the chief risk in failing to address impact - has been recognised for decades (Pressey, 1994; Runte, 1979, ch. 3: Worthless lands). The first high-profile call to consider impact in conservation (Ferraro and Pattanayak, 2006) is now a decade old, and attempts to estimate impact of protected areas began earlier (reviewed by Gaston et al., 2008). Framing policy targets and operational objectives in terms of impact is tractable (Pressey et al., 2015). The barrier to focusing on impact is attachment to measures that are familiar, easy to estimate, scientifically rewarding, and politically convenient, even if they are misleading.

For those who want to use them, methods are established to estimate conservation impact retrospectively (Ferraro and Hanauer, 2014) and predictively (Fulton et al., 2015; Visconti et al., 2015). The methods vary in expense and rigour (Margoluis et al., 2009), and the most rigorous might be reserved for a representative sample of conservation actions and socio-economic contexts to support decisions more broadly. Retrospective assessments yield lessons but, ultimately, predicting impact is essential if we are to make more difference in the future, and prediction inevitably involves uncertainty. One choice for decision-makers is therefore: stay with measures that can be estimated accurately but might bear no relationship to impact, or deal with the uncertainty inherent in making a difference. The outlook for what remains of the planet's biodiversity will be brighter if we move to the second option. This means substituting beliefs in unreliable measures with an evidence base for effective actions.

5. Levels of counterfactual evidence for policy, planning, and management

Conservation lags well behind medicine in its use of evidence (Sutherland et al., 2004), even though their missions are similar. Medicine's mission is to save and improve human lives. One of conservation's primary missions is to save biodiversity, although in a more complex and contested context than medicine's (Fazey et al., 2004). Like other crisis disciplines, conservation must balance opposing risks: decisions are necessary without all the necessary information to avoid delays that could aggravate loss of biodiversity (Soulé, 1985), while wrong decisions lead to loss that could have been avoided (Table 1, and see Pullin and Knight, 2009). Reducing both kinds of risk requires decision-makers to review belief systems in the light of evidence.

Scientific evidence is, of course, just part of the larger picture of decision-making. It is a complement to, not replacement for, experience (Fazey et al., 2004), and it contributes to decisions made amid the additional influences of competing social, political, and financial interests. At best, then, many conservation decisions are evidence-informed, not evidence-based, and the purpose of conservation science is to assemble the evidence (Haddaway and Pullin, 2013), albeit by defining problems more tightly specified than many real-world challenges (Adams and Sandbrook, 2013).

The role of evidence in conservation is to test, modify, and inform decision-makers' models of the world. All decision-makers use models –

implicit or otherwise - to evaluate alternative actions by anticipating their results. With positive results are defined in terms of conservation impact, models for decisions can be misleading in three ways:

1. Directed by mistake to goals other than impact, as when extent or representativeness are confused with conservation progress;
2. Directed intentionally to goals other than impact. The motivations

Table 2

Levels of counterfactual evidence for conservation decision-making. The examples of each level are ways of identifying spatial priorities for conservation actions (ways of predicting how limited conservation resources can be most effectively allocated). The same levels could be applied to formulating policy targets and operational objectives, and gauging progress towards targets and objectives. Ways of identifying priorities include both direct delineation of priority areas and aspects of decision-making processes that lead to delineation. At each level are examples of approaches to defining conservation priorities, with outlines of their limitations. Generic limitations refer to the ability of analyses to account for the real-world contexts of conservation decisions (e.g. threats to biodiversity, opportunities for and constraints on actions, uncertainties in data, political pragmatism, preferences of stakeholders). Shading indicates no counterfactual evidence or no evidence for avoided threats or avoided loss of biodiversity.

Level of counterfactual evidence	Generic limitations	Comments regarding impact
Level 1A: Opinions. Opinions of individuals or groups about priority areas or ways of identifying them, without the structured, explicit methods for elicitation used for Level 3A Examples: 1. Assumption: larger inputs necessarily produce better results for biodiversity 2. Assumption: large, remote areas are priorities for protection (e.g. Mittermeier et al. 2003) 3. Best–practice recommendations not supported by comparative analyses (e.g. separating land–use pressures from derivation of objectives, Knight et al. 2006b) 4. Assumption: boundary rationalisation for protected areas should be prioritised over new reserves 5. Expert–derived conservation priorities drawn on maps	No explicit testing of effectiveness of assumptions or recommendations in achieving stated objectives (if any) in the face of real–world constraints and opportunities. Individuals have limited context for opinions; training and experience do not necessarily equip individuals to adequately address new situations or innovative goals. Tendency toward groupthink in research groups and organisations (when the desire for harmony within a group overrides a realistic appraisal of alternatives, Martin et al. 2012). Resistance by individuals and organisations to ideas and approaches that threaten established ones.	No explicit basis for reliable estimates of counterfactual conditions in the absence of actions.
Level 1B: Criteria–based methods. Use of one or more explicit criteria to identify areas important for conservation actions, essentially formalising opinions with qualitative or quantitative rules Examples: 6. Protected–area management effectiveness (Leverington et al. 2010) as a guide to priorities for management resources 7. Identifying Key Biodiversity Areas (Foster et al. 2012) 8. Identifying Ecologically or Biologically Significant Marine Areas (EBSAs) (Convention on Biological Diversity 2016)	Criteria are ways of encouraging explicitness and consistency in methods. Criteria do not constitute explicit testing of effectiveness of assumptions or recommendations in achieving stated objectives (if any) in the face of real–world constraints and opportunities.	No explicit basis for reliable estimates of counterfactual conditions in the absence of actions.
Level 2: Static spatial comparisons of representation. Testing of approaches to identifying priorities based on spatial configuration of resulting conservation areas and attributes of those areas. Examples: 9. Spatial configurations and total costs of priority areas (Carwardine et al. 2008) 10. Priorities for increasing equality of representation of ecoregions within countries (Barr et al. 2011) 11. Testing limitations of biodiversity surrogates used in conservation planning (Grantham et al. 2010)	Combine diverse and non–equivalent attributes of areas (e.g. spatial configuration and multiple aspects of costs, threats, and feasibility of conservation actions) in essentially arbitrary ways. Evidence is particularly weak when the case for using a certain type of input data rests on comparing results with and without the same input data (e.g. Carwardine et al. 2008). Constitute just one part of a much larger planning process (Pressey and Bottrill 2009), so even optimal results for defined objectives could lead to poor results in the larger planning context. Even in rare situations allowing short–term implementation of all identified areas, initial outputs from conservation planning software are altered significantly to accommodate preferences of stakeholders (Pressey et al. 2013).	Potential for estimates of counterfactual conditions in the absence of actions, but the resulting impact (larger outcomes with actions than without) is uninformative about avoided threats and avoided loss of biodiversity.

Level 3A: Expert–elicited comparisons. Testing of approaches to avoiding threats or avoiding loss of biodiversity with explicit organisation of expert knowledge. Examples are structured decision–making (Martin et al. 2009) and elicitation methods that combine multiple judgments, minimise bias, and verify accuracy (Martin et al. 2012). Examples: 12. Identification of management actions to mitigate main threats to biodiversity (Chadès et al. 2014) 13. Expected benefits of management actions to regional biodiversity, with and without climate change (Firn et al. 2015) 14. Expected effects of disturbance on mortality and sub–lethal injury of animals (Fleishman et al. 2016)	The full complexity of dynamic physical, ecological, and socio economic variables is essentially impossible to consider in these predictions, which necessarily address simplified versions of conservation problems. To some extent, this limitation can be explored with sensitivity analysis relating to uncertainty and causal factors considered.	If not already used to estimate counterfactual conditions, then easily adapted to do so.
Level 3B: Dynamic spatial comparisons. Testing of approaches by simulating aspects of implementation dynamics, including expanding threats and the interactions between conservation actions and parallel and/or resultant human–induced changes to ecosystems. These tests vary in attributes of threats, biodiversity, and human–induced changes considered, and the types of interactions between them. An initial set of categories is outlined below, with examples.	With increasing complexity (3B.1 to 3B.4) these models attempt to account for more aspects of the planning process described by Pressey and Bottrill (2009), including preferences of stakeholders and their responses to conservation actions. However, the full complexity of dynamic physical, ecological, and socioeconomic variables is essentially impossible to model, although parts of it can be explored with scenarios and sensitivity analysis.	
3B.1 Threat–only models Example: 15. Expected deforestation (Sloan and Pelletier 2012)	Spatially and temporally explicit models of expanding, intensifying, or new threats are always subject to error. Consequences of errors can be explored with sensitivity analysis. Large uncertainties might justify ignoring threats in prioritisation (Visconti et al. 2010a)	Basis for estimating counterfactual conditions so that conservation impact (avoided threats and/or avoided loss of biodiversity) can be predicted. Avoided threats might be the most reliable estimate of potential conservation impact where data on biodiversity are highly generalised and uncertain (e.g. marine ecoregions).
3B.2 Simple dynamic interaction models Example: 16. Interacting conservation and loss of biodiversity pattern, such as mapped types of native vegetation, with progressively updated priorities (Pressey et al. 2004)	Models of threats subject to error, as in 3B.1. Models of biodiversity distribution and responses to threats also subject to error. Consequences of errors can be explored with sensitivity analysis.	If not already used to estimate counterfactual conditions, then easily adapted to do so. Add information to models in 3B.1 by predicting effects of threats on elements of biodiversity pattern, providing estimates of avoided loss.
3B.3 Dynamic interaction models with ecosystem or human responses Examples: 17. Interactions between conservation actions and threats and their influence on biodiversity processes related to persistence of species and composition of ecosystems (Gurney et al. 2013; Visconti et al. 2010b) 18. Interactions between conservation actions and fishers, considering larval spillover from reserves and displacement of fishing effort (Brown et al. 2015)	Models of threats subject to error, as in 3B.1. Models of biodiversity distribution and responses to threats also subject to error, as in 3B.2. Error involved in modelling of additional biodiversity variables and, particularly, human responses, which are likely to be idiosyncratic and space and time. Consequences of errors can be explored with and sensitivity analysis.	If not already used to estimate counterfactual conditions, then easily adapted to do so. Add information to models in 3B.2 by considering selected biodiversity processes or human responses to conservation actions and their implications for avoided loss.

3B.4 Dynamic interaction models with ecosystem and human responses. Simulation models involving dynamics of threats, simple to complex ecosystem responses to threats, and diverse human responses to conservation actions Example: 19. A suite of models described by Fulton et al. (2015), considering interacting species, responses of multiple users of marine resources, including displacement and non–compliance, with applications to policy formulation.	Models of threats subject to error, as in 3B.1. Models of biodiversity distribution and responses to threats also subject to error, as in 3B.2. Error involved in modelling of additional variables, particularly human responses, which are likely to be idiosyncratic in time and space. Consequences of error can be explored with sensitivity analysis.	Some applications have estimated counterfactual conditions. Others easily adapted to do so. Add information to models in 3B.3 by considering interactions between ecosystem dynamics and human responses to conservation actions and their implications for avoided loss.
Level 4: Complex–system models. Extending level 3 methods in several ways: • Testing approaches in the context of the whole process of conservation planning (Pressey and Bottrill 2009), including: constraints on budget and time–frame; preferences of diverse stakeholders; limitations of data and quantitative objectives; dynamics of implementation; and limitations of ongoing maintenance and monitoring of established conservation areas. • Recognising that conservation decision–making operates in socioecological contexts with characteristics of complex systems, including (Game et al. 2014): numerous interacting elements lacking central control nonlinear interactions between elements; constant change, only partly attributable to conservation actions; effects of actions that cannot be fully predicted; and inevitable tradeoffs between diverse responses to actions. • Closing the adaptive management loop, in which assumptions are revisited and actions re–evaluated as decision–makers respond to new information and changing physical and ecological conditions, some of which result from previous decisions, and reactions of stakeholders (Polasky et al. 2011). • Linking models to account for: ultimate pressures on biodiversity, including demand for and supply of different commodities; proximate pressures such as harvesting, habitat loss, and climate change; local actions to protect and restore biodiversity; and ecological and human responses to actions.	Although this comprehensive analysis of decision–making would yield new insights (including the limitations of the more simplified quantitative testing above), this level of analysis has apparently not yet been attempted.	Potential applications to estimate counterfactual conditions and the influence of different parts of the planning process on conservation impact. Potential to draw lessons from complexity science applied to other fields, including the benefits of distributed decision–making, with clear, overall objectives and principles established to allow context–specific decisions by suitably skilled people with good local knowledge.

might be:
- cynical, as when governments seek to impress electorates with the sheer extent of protected areas; or
- well-meaning but unfounded, as when conservationists believe, often with limited evidence, that easy wins in residual places will set the scene for more meaningful actions in future.

3. Directed towards impact, but involving invalid assumptions about how this might be achieved or based on inaccurate parameters, such as expected rates and patterns of biodiversity loss in the absence of actions.

Decision-makers' models of the world are necessarily rooted in experience and intuition but, if not examined explicitly and critically, can lead to poor results for biodiversity (Addison et al., 2013). Models can be made explicit with theories of change that map out the presumed causal links, direct and indirect, between actions and consequences (Ferraro and Hanauer, 2015; Margoluis et al., 2013). In turn, theories of change can be refined into expert-elicited predictions or formal simulations of how decisions might play out in space and time in different biophysical and socioeconomic contexts. Expert-elicited predictions can specify the extent and intensity of threats and their effects on biodiversity, and the relative effectiveness of conservation actions in mitigating those threats (Chadès et al., 2014; Firn et al., 2015; Fleishman et al., 2016). The term “simulations” here includes spatial modelling projected through time and structured decision-making based on scenarios (Milner-Gulland et al., 2010; Polasky et al., 2011). Simulations can be fully quantitative (Fulton et al., 2015; Visconti et al., 2015) or involve elements of expert judgement and qualitative assessment (Margoluis et al., 2009; Polasky et al., 2011). Expert-elicited predictions and formal simulations remain models, but they have important advantages over the implicit models in managers' heads. Their explicitness means they can be scrutinised and refined, subjected to sensitivity analysis to identify dependencies on uncertain parameters (Pressey et al., 2004), used to explore interactions between pressures on biodiversity, including climate change (Gurney et al., 2013), and adapted to investigate options for policy and practice (Fulton et al., 2015). Launching policies on taxation or social welfare without explicit economic modelling would be seen as irresponsible. The same attitude should apply to promoting conservation priorities without modelling their conservation impact.

The framework in Table 2, as a starting point for refinement, concerns levels of counterfactual evidence to test the potential impact of conservation decisions. The framework complements other forms of evidence, such as systematic reviews (Pullin and Knight, 2009). For brevity, the scope is limited here to decisions about spatial priorities. In this context, the purpose of counterfactual evidence is to estimate future conditions with and without conservation actions taken with the approach being tested. The rationale is straightforward:

- Conservation priorities are no more than predictions, usually untested, about the best ways to spend limited conservation resources

Table 3
Types of measures in relation to levels of counterfactual evidence, using the examples (numbered) from Table 2. Shaded cells indicate the intersection of measures with counterfactual conditions of interest for testing conservation priorities (columns) and methods capable of estimating these conditions (row).

	Types of measures				
Level of evidence	Inputs	Outputs	Outcomes (representation)	Outcomes (threats)	Outcomes (biodiversity)
1A. Opinions	1	2			3,4,5
1B. Criteria–based	6				7,8
2. Static spatial comparisons			9,10,11		
3A. Expert–elicited comparisons				12,13	12,13,14
3B. Dynamic spatial comparisons				15	16,17,18,19
4. Complex–systems models			Aspirational		

1 – Assumption: larger inputs necessarily produce better results for biodiversity; 2 – Assumption: large, remote areas are priorities for protection; 3 – Best–practice recommendations not supported by comparative analyses; 4 – Assumption: –boundary rationalisation for protected areas should be prioritised over new reserves; 5 – Expert–derived conservation priorities drawn on maps; 6 – Protected–area management effectiveness; 7 – Identifying Key Biodiversity Areas; 8 – Identifying Ecologically or Biologically Significant Marine Areas; 9 – Spatial configurations and total costs of priority areas; 10 – Priorities for increasing equality of representation; 11 – Testing limitations of biodiversity surrogates used in conservation planning; 12 – Identification of management actions to mitigate threats; 13 – Expected benefits of management actions to biodiversity; 14 – Expected effects of disturbance on animals; 15 – Threat–only models; 16 – Simple dynamic interaction models; 17 – Dynamic interaction models with ecosystem responses; 18 – Dynamic interaction models with human responses; 19 – Dynamic interaction models with ecosystem and human responses

- Rigorous testing of these statements is warranted by the risks to biodiversity of getting them wrong, and by the resources that could be wasted when mistakes are made
- The questions to be addressed by testing are:
  - o What conservation impact is expected to result from the recommended approach?
  - o Would one or more alternative approaches have more impact?

In Table 2, opinions (level 1A)), such as large, remote areas necessarily being priorities for conservation (Mittermeier et al., 2003), do not constitute counterfactual evidence, and are least reliable when approaches are recommended without understanding how they will manifest on the ground (e.g. the recommendation to ignore threats when setting objectives, Knight et al., 2006b). Criteria-based methods (level 1B) do not constitute counterfactual evidence. Criteria promote explicit, consistent definition of priority areas but cannot test those priorities in terms of their potential impact. Persuasion, groupthink (Martin et al., 2012), and even extensive consultation and international endorsement, in the case of Key Biodiversity Areas (Foster et al., 2012) or Ecologically or Biologically Significant Marine Areas (Convention on Biological Diversity, 2016), are not replacements for evidence. Representation analyses (level 2), either for planning (Carwardine et al., 2008) or testing of biodiversity surrogates (Grantham et al., 2010), provide only superficial insights into impact: predicting more features represented with conservation areas than without raises the question of whether loss will be avoided by protecting features most in need of protection. The state of the art in counterfactual evidence is defined by expert-elicited comparisons using rigorous methods (level 3A), such as structured decision-making (Martin et al., 2009), and dynamic spatial simulations of varying complexity (level 3B), including models of threats (Sloan and Pelletier, 2012; Visconti et al., 2010a) and their interactions with biodiversity (Brown et al., 2015). These approaches merge to the extent that expert knowledge is incorporated into spatially and temporally explicit simulations. Both sets of approaches can provide insights into potential impact by predicting conditions with and without conservation action. Complex-systems models (level 4) are aspirational but encompass methods regarded by experts in simulations as tractable and necessary for understanding conservation impact.

Four main points emerge from the assessment in Table 2. First, shaded cells in the summary table (Table 3) indicate those parts most relevant to testing approaches to prioritisation: the intersection of (columns) measures that have counterfactual conditions of interest for understanding impact and (rows) methods capable of estimating those counterfactual conditions. Testing for impact, including testing decision-makers' implicit models and predictions at levels 1 or 2, therefore begins at level 3. Second, numerical sophistication in defining priorities, if it lacks estimation of counterfactual conditions, does not constitute evidence of effectiveness. This applies to criteria-based approaches, but also to quantitative methods from systematic conservation planning. The combination of irreplaceability and vulnerability used to define priority by Margules and Pressey (2000) was an interpretation of opinion (level 1A), expressed through quantitative criteria (level 1B). It was an untested prediction before the basic simulations of Pressey et al. (2004, level 3B.2), the conclusions from which have now been modified by subsequent work (Visconti et al. 2010b, level 3B.3). Third, the complexity of information that prioritisation measures attempt to provide (Table 3 columns) is not necessarily related to the levels of evidence (Table 3 rows) that support their use. For example, outcomes for biodiversity – complex interplays of physical, ecological, and socio-economic dynamics - have been predicted by untested opinions and criteria.

Putting aside opinions and criteria, which provide little scope for exploring uncertainties, a fourth main point from Table 2 is that the reliability of models for estimating counterfactuals might not be related to their complexity. So simulation models (level 3B) are not necessarily more accurate than expert-elicited predictions at level 3A (Fleishman et al., 2016), and the progression in complexity of simulations from 3B.1 to 3B.4 is not necessarily accompanied by more reliability, once input data and model errors are considered. A challenge for conservation planners is therefore to better understand how alternative approaches balance uncertainty in predictions against attempts at complexity and realism. It will be important to consider, for example, the extent to which complex-systems models (level 4) are constrained by lack of data as well as the difficulty and uncertainty involved in assembling the models themselves.

Spatially and temporally explicit simulations, and the theories of change that underpin them, are important missing elements in the evidence base for conservation decisions. Simulations extend systematic methods (which required, for the first time in the 1980s, decision-makers to state explicitly what they intended to achieve) by also requiring statements of core assumptions and parameters for threats, their potential effects on biodiversity, and the mitigating influences of actions. What if the required data and skills to build simulations are unavailable in some regions, or model uncertainties are likely to be very large? In these cases, even explicit theories of change directed towards impact would help to expose decision-makers' belief systems to the scrutiny they need (Ferraro and Hanauer, 2015), just as the structured process of planning can be as important as specific software outputs (Wendt et al., 2016). A viable fallback from formal simulations (level 3B) could also be expert-elicited predictions (level 3A). One way of structuring these predictions is through theories of change with causal links given quantitative values and, ideally, uncertainties made explicit, exemplified by Bayesian belief networks of the effects of management actions

(Ban et al., 2015).

Both theories of change and simulations directed at conservation impact are warranted by the serious risks to biodiversity of decisions based on unexamined belief systems (Table 1). Theories of change are feasible for any organisation and all practitioners. Predictions based on expert knowledge or simulations require a budget line, but this is minor for large organisations, and these approaches are technically tractable at varying extents and resolutions, from global (Visconti et al., 2015) to local (Visconti et al., 2010b). Level-3 predictions can be developed for a representative sample of actions in representative physical, ecological, and socioeconomic contexts. Along with a more general move to evidence in conservation, increased use of theories of change, expert predictions, and simulations will encounter and overcome difficulties, but the return on investment will be increased confidence and accountability in decision-making.

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