Conservation Biology

Essay

Reconciling multiple counterfactuals when evaluating biodiversity conservation impact in social-ecological systems

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Abstract: When evaluating the impact of a biodiversity conservation intervention, a counterfactual is typically needed. Counterfactuals are possible alternative system trajectories in the absence of an intervention. Comparing observed outcomes against the chosen counterfactual allows the impact (change attributable to the intervention) to be determined. Because counterfactuals by definition never occur, they must be estimated. Sometimes, there may be many plausible counterfactuals, including various drivers of biodiversity change and defined on a range of spatial or temporal scales. Here, we posit that, by definition, conservation interventions always take place in social-ecological systems (SES) (i.e., ecological systems integrated with human actors). Evaluating the impact of an intervention in an SES, therefore, means taking into account the counterfactuals assumed by different human actors. Use of different counterfactuals by different actors will give rise to perceived differences in the impacts of interventions, which may lead to disagreement about its success or the effectiveness of the underlying approach. Despite that there are biophysical biodiversity trends, it is often true that no single counterfactual is definitively the right one for conservation assessment, so multiple evaluations of intervention efficacy could be considered justifiable. Therefore, we propose calculating the sum of perceived differences, which captures the range of impact estimates associated with different actors in a given SES. The sum of perceived differences gives some indication of how closely actors in an SES agree on the impacts of an intervention. We applied the concept of perceived differences to a set of global, national, and regional case studies (e.g., global realization of Aichi Target 11 for marine protected areas, effect of biodiversity offsetting on vegetation condition in Australia, and influence of conservation measures on an endangered ungulate in Central Asia). We explored approaches for minimizing the sum, including a combination of negotiation and structured decision making, careful alignment of expectations on scope and measurement, and explicit recognition of any intractable differences between stakeholders.

Keywords: baseline, conservation impact, impact evaluation, reference frame

Reconciliación de Múltiples Hipótesis de Contraste al Evaluar el Impacto de la Conservación de la Biodiversidad en los Sistemas Socio-Ecológicos

Resumen: Cuando se evalúa el impacto de una intervención de conservación de la biodiversidad, generalmente se requiere una hipótesis de contraste. Las hipótesis de contraste son las posibles trayectorias alternativas del sistema en ausencia de una intervención. La comparación de los resultados observados con la hipótesis de contraste elegida permite que se determine el impacto (cambio atribuible a la intervención). Ya que las hipótesis de contraste por definición nunca ocurren, éstas deben ser estimadas. En algunos casos es posible que existan

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muchas hipótesis de contraste, incluyendo a muchos conductores del cambio en la biodiversidad, y que estén definidas bajo una gama de escalas espaciales o temporales. En este artículo planteamos que, por definición, las intervenciones de conservación siempre ocurren en sistemas socioecológicos (SES) (es decir, sistemas ecológicos integrados con actores humanos). Por lo tanto, la evaluación del impacto de una intervención en un SES implica la consideración de las hipótesis de contraste asumidas por los diferentes actores humanos. El uso de diferentes hipótesis de contraste por los diferentes actores hará que surjan diferencias percibidas en los impactos de las intervenciones, lo que puede llegar a discrepancias sobre su éxito o sobre la efectividad de la estrategia subyacente. A pesar de que existen tendencias biofísicas de la biodiversidad, con frecuencia es cierto que no hay una sola hipótesis de contraste que sea correcta de manera definitiva para la evaluación de la conservación, por lo que múltiples evaluaciones de la eficiencia de la intervención podrían considerarse como justificables. Así, proponemos que se calcule la suma de las diferencias percibidas, la cual captura la gama de las estimaciones de impacto asociadas con diferentes actores en un SES dado. La suma de las diferencias percibidas nos da algún tipo de indicación sobre cuán de acuerdo están los actores de un SES sobre los impactos de una intervención. Aplicamos el concepto de diferencias percibidas a un conjunto de estudios de caso mundiales, nacionales y regionales (p. ej.: la realización mundial del Objetivo Aichi 11 para las áreas marinas protegidas, el efecto de la compensación de la biodiversidad sobre las condiciones botánicas en Australia y la influencia de las medidas de conservación sobre un ungulado en peligro en Asia central). Exploramos las estrategias para minimizar la suma, incluyendo una combinación de negociación y toma estructurada de decisiones, la alineación cuidadosa de las expectativas sobre el enfoque y la medida y el reconocimiento explícito de cualquier diferencia intratable entre los actores sociales.

Palabras Clave: evaluación del impacto, impacto de la conservación, línea base, marco de referencia

摘要: 在评估生物多样性保护干预措施的影响时, 通常需要一个反设事实。反设事实是在没有干预的情况下可能的替代性系统轨迹。将观察到的结果与选择的反设事实进行比较, 则可以确定影响 (干预引起的变化) 。由于从定义上讲, 反设事实永远不会发生, 因此必须对其进行估计。然而, 有时可能存在许多看似合理的反设事实, 包括生物多样性变化的各种驱动因素, 并且可以在不同的时空尺度范围内加以定义。本研究假定, 根据定义, 保护干预总是发生在社会生态系统中 (即生态系统与人类行为相结合) 。因此, 评估一项干预措施对社会生态系统的影响需要考虑不同人类行为的反设事实。针对不同行动者使用不同的反设事实, 会导致干预措施的影响也出现明显差异, 这可能使人们对干预成功与否或方法有效性的评估产生分歧。虽然存在生物物理上的生物多样性趋势, 但往往没有单一的反设事实在保护评估中绝对适用, 因此应当对干预效果进行多重评估。我们建议计算可观测的差异的总和, 这样可以包含既定社会生态系统中不同行动者的影响估计范围。观测差异的总和还可以在一定程度上表明社会生态系统中的行动者对干预影响的一致性。我们将观测差异的概念应用于一系列全球、国家和区域的案例研究中 (如爱知目标 11 中海洋保护区的全球实现情况、澳大利亚生物多样性补偿对植被的影响, 以及中亚的保护措施对一种濒危有蹄类动物的影响)。本研究进一步探索了使观测总差异最小化的方法, 包括结合协商与结构化决策, 仔细调整对影响范围和测定结果的期望, 以及清楚认识到利益相关者之间难以处理的差异。**【翻译: 胡怡思; 审校: 聂永刚】**

关键词: 基线, 保护影响, 影响评估, 参考架构

Introduction

The effectiveness of attempts to conserve biodiversity–and its associated contributions to human well-being–has become an increasingly pressing topic. A counterfactual is necessary to quantitatively evaluate the ecological impact of a biodiversity conservation intervention (Ferraro & Hanauer 2014). Counterfactuals are a type of reference scenario that captures an alternative possible trajectory of a dynamic system in the absence of a given intervention. The impact of the intervention is the change attributable to the intervention, measured as the difference between the actual observed trajectory (the outcome) and the predicted counterfactual (Ferraro & Pattanayak 2006). Numerous counterfactuals can be reasonably specified for most systems relevant to conservation because it is possible to select from a range of drivers of system change for potential inclusion in the counterfactual (Maron et al. 2018), notwithstanding that counterfactuals can also be specified over various spatiotemporal scales (Bull et al. 2014). Crucially counterfactuals are, by definition, a scenario that does not occur, so they can never be directly observed and monitored–and there is often no single correct counterfactual; rather, there are various counterfactuals of differing plausibility. This is even true to an extent for control sites used in quasi-experimental methods because subjective decisions have to be made when choosing such sites to reduce the impacts of confounding factors (e.g., Wiik et al. 2019). Thus, although such control sites may give a good approximation to what would have happened at the treatment sites without the intervention, they are still open to some interpretation.

Typically, conservationists are interested in a counterfactual representing the alternative ecological trajectory of a system, which is often influenced by multiple anthropogenic activities beyond the intervention in question (Ferraro & Pattanayak 2006; Maron et al. 2018). But–whereas the literature on impact evaluation is accumulating rapidly, along with tools for implementation–impact evaluation is still rarely carried out in practice (Wiik et al. 2019). Moreover, conservation interventions do not take place in purely ecological systems; they take place in social-ecological systems (SES) (Berkes & Folke 1998) (i.e., ecological systems integrated with social systems consisting of human actors). Thus, it is critical to go beyond purely ecological counterfactuals when evaluating the impact of a conservation intervention and consider interlinked social systems (Maron et al. 2018). This adds considerable challenges because different actors may subjectively assume different counterfactuals are most relevant when judging impact (e.g., whether to use a counterfactual at the spatial scale of the project or the landscape [Bull et al. 2014]) due to factors, such as unconscious biases (Tversky & Kahneman 1974), temporal starting point (Pauly 1995), or assumptions made about processes driving change. This is important because the choice of counterfactual not only alters perceptions about intervention success, but also potentially the actual actions of stakeholders (Bull et al. 2014). Therefore, it is insufficient to consider counterfactuals on a purely ecological basis when judging conservation impact. Also deserving attention are the ecological counterfactuals associated with differing interpretations from relevant stakeholders in the SES. We term this set of possible social-ecological counterfactuals (i.e., ecological counterfactuals derived from varying stakeholder perceptions) a family of counterfactuals.

Our objective was to further formalize the use of counterfactuals for evaluating biodiversity outcomes in an SES. Biodiversity is not the only property of an SES that might require conservation interventions, but it is our focus. We developed an exploratory conceptual framework, illustrated (although not formally tested) with case studies. In particular, we focused on differences of interpretation during quantitative evaluation of the ecological impact of conservation interventions. Approaches for qualitative evaluation exist (Sutherland et al. 2018). However, because they do not require quantitative impact estimates, we did not consider them. We did, though, explore how an actor's precise choice of counterfactual arises from the actor's personal reference frame. We then explored approaches that minimize divergence in personal reference frames and thus the choice of counterfactual to avoid conflict between actors' perceptions. Ultimately, we aimed to improve impact evaluation through consideration of not only biophysical outcomes relative to counterfactuals, but also of how multiple stakeholders view the plausibility of different counterfactuals.

Toward Families of Counterfactuals

In a hypothetical ecosystem, we considered conservation interventions to be targeted at biodiversity, but framed more broadly around "people and nature" (emphasizing "the importance of cultural structures and institutions for developing sustainable and resilient interactions between human societies and the natural environment" [Mace 2014]). By definition, an intervention being implemented means that: at least 1 human actor has the potential to influence that biodiversity; at least 1 human actor must be affected by that influence on biodiversity; and at least 1 human actor must be responsible for the intervention. In the simplest case, these situations describe the same actor. For instance, if the SES is a farm, the single actor might be a farmer creating habitat for declining bird species. The logic is that for any system in which a conservation intervention is taking place, there must be at least 1 relevant human actor–that is, every system involving a conservation intervention must be an SES.

We focused on social perceptions of the ecological impact of an intervention, a necessary precursor to evaluating its social impacts. Although social impacts require similar treatment (e.g., Davidson 2013), they are beyond the scope of this article. So, to evaluate the impact of the intervention, we specified a counterfactual ("a causal effect of a program is only defined with respect to a well-defined alternative" [Ferraro & Hanauer 2014]). The actor anticipates an alternative biodiversity trend that would have taken place without the intervention. But, biodiversity measurement is open to interpretation (Purvis & Hector 2000), and subcomponents of biodiversity are often ascribed wildly different weightings depending on the actor (Baylis et al. 2016; Bull & Maron 2016; Pearson 2016). Compounding this is the difficulty of determining which components of biodiversity are relevant when going beyond static considerations–which depends on the chosen spatiotemporal scale (Bull et al. 2014). Finally, the shape of the counterfactual trend will be heavily influenced by an actor's expectations (Ferraro & Hanauer 2014). Yet, in the simplest case of our hypothetical SES, the single actor is free to select whatever counterfactual this actor chooses, and the perceived impact of the intervention is the difference between the observed biophysical outcome and that counterfactual (Ferraro & Hanauer 2014). Continuing our farmer example, a farmer might compare outcomes with the counterfactual scenario in which no new habitat was created (acknowledging multiple plausible futures).

A more complex hypothetical scenario with 2 actors would involve 1 actor carrying out activities that reduce biodiversity and the other implementing conservation interventions. Assuming it is relevant for both actors to

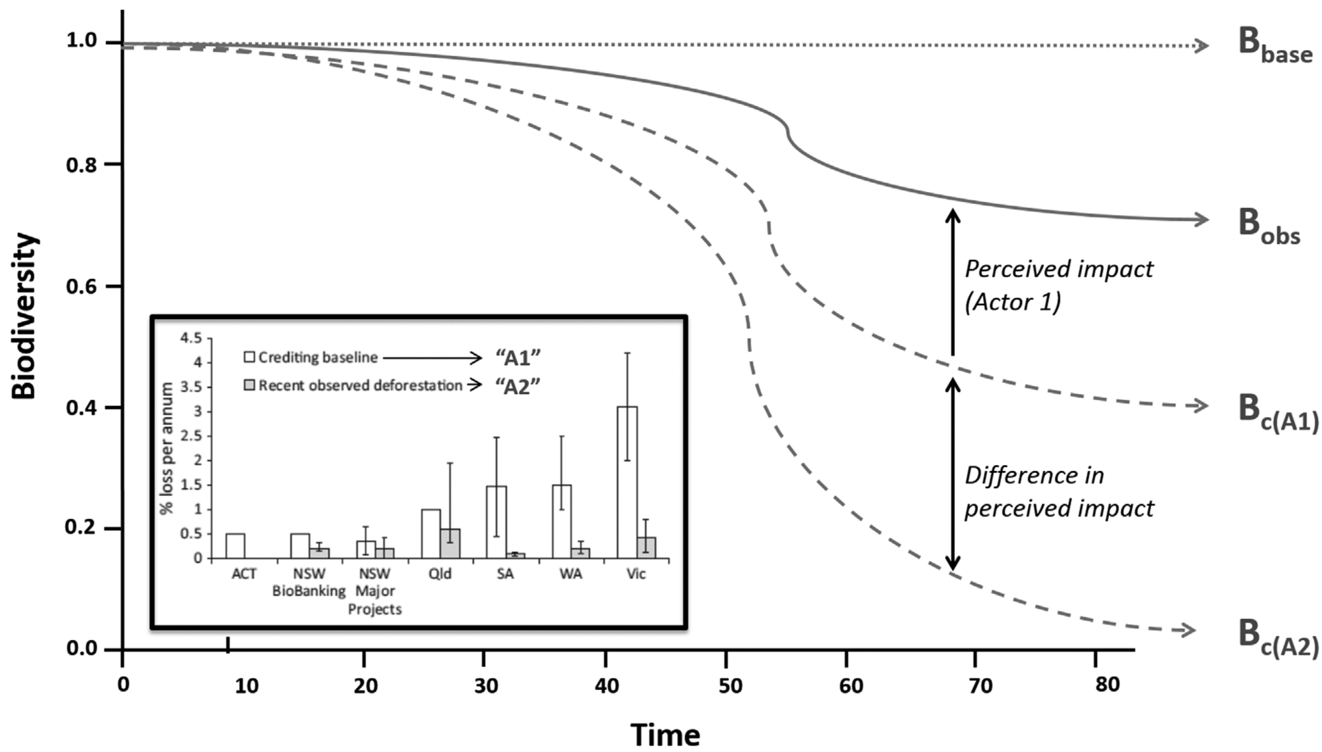

Figure 1. Hypothetical differences in counterfactual biodiversity trends ($B_{c[A1]}$, $B_{c[A2]}$) used by 2 actors (A1 and A2) in a hypothetical social-ecological system compared with a baseline (B_{base}) at time 0 and the observed biodiversity outcomes through time (B_{obs}). Actor 1's perceived impact is based on a comparison of B_{obs} with $B_{c(A1)}$, whereas actor 2's is based on a comparison of B_{obs} with $B_{c(A2)}$. The difference in perceived impact is shown. Inset: real-life example of 2 actors judging different counterfactual deforestation rates in Australian states (from Maron et al. 2015) (abbreviations defined in Table 1).

monitor impact, each specifies a counterfactual and measures the outcome. There is a wealth of reasons why their choice of counterfactual might differ, but even if their choice is the same in theory, expectation or uncertainty may mean the precise trajectory of the chosen counterfactual diverges. The perceived impact for actor 1 is the difference between the observed biophysical outcome and actor 1's chosen counterfactual, and for actor 2 it is the difference between the same biophysical outcome and actor 2's chosen counterfactual. If the counterfactuals are different, there is consequently a difference between perceived impact for actor 1 and actor 2, which–whatever the actual trajectory–equals the difference in their chosen counterfactuals (assuming they used the same measure of outcome) (Fig. 1 & Supporting Information). For a real example, see Maron et al. (2015), who found Australian policy makers (actor 1) using different counterfactual rates from those calculated by researchers (actor 2) for biodiversity offset conservation interventions (Fig. 1 and below). Analogously, multiple reference levels are used by different actors for climate-change mitigation (Griscom et al. 2009).

Finally, consider the general case of 3 or more actors in the hypothetical SES who all evaluate the impact of an intervention, but at least some specify different counterfactuals. This resultant set of counterfactuals is the aforementioned family of counterfactuals. For the SES as a whole, consider the total difference in perceived impact across the family of counterfactuals for the full diversity of stakeholders. The larger the total difference is, the greater the difference in perceived impact for multiple actors, which may imply that some actors are not satisfied with the intervention. We propose calculating a sum of perceived differences for the family of counterfactuals, which is the sum of the magnitude of the difference between the counterfactuals assumed by every actor and every other actor in the system (see Supporting Information for mathematical formulation and justification). The sum could conceivably be used to weight the counterfactuals assumed by various individuals differently, for example, to account for different uncertainties or uneven power dynamics (see "Minimizing Perceived Differences"). Because no counterfactual is definitively correct, rather, counterfactuals are chosen on the basis of actors' value judgments, the sum of perceived differences is necessary to capture the impact of a conservation intervention in an SES. Importantly, the sum incorporates the counterfactuals actually used by actors without differentiating between those counterfactuals that are more or less plausible. Applying this framework makes the counterfactuals used more transparent and thus facilitates discussion around plausibility.

Actual biodiversity trends are, in principle, objective and biophysical. Yet, because the result of conservation interventions will be scrutinized by many different actors with their own personal counterfactuals, which will often diverge, even robustly monitored interventions could lead to wide-ranging and at times conflicting interpretations of efficacy (Pearson 2016). Equally, because robust evaluation of interventions must be linked inextricably to the initial design of the interventions themselves (Bull et al. 2014), the challenge of divergence in counterfactuals is important when setting conservation objectives. Concerning the sustainability of interventions, it would be preferable to minimize the sum of perceived differences for an SES–implying that most actors use approximately the same counterfactual. To determine whether it is possible to reduce the sum of perceived differences, we first considered the reference frame in which the counterfactual is specified.

Counterfactuals and Reference Frames

Again, counterfactuals are a type of reference scenario (Maron et al. 2018). Reference scenarios are specified in a reference frame (or frame of reference), and any number of reference scenarios can be specified in 1 reference frame (Bull et al. 2014). In Supporting Information, we formally describe a reference frame and discuss use of the term in different scientific disciplines. To a greater or lesser extent across different disciplines, a reference frame is typically composed of the relevant parameter space that captures all possible variables of interest, particularly the subset of interest to the relevant observer and the observers themselves, including the social values through which an observer views the chosen parameters. Unlike other disciplines in the natural sciences, which do so alongside rather than as part of the reference frame, we also included the specific coordinate system used to make measurements in the parameter space, chosen by the observer, which determines the spatiotemporal scale of the reference frame (Supporting Information). A coordinate system is a set of numbers used to specify quantitative information (e.g., an object's location in space [Supporting Information]). In contrast to measurement of physical change in disciplines where standardized units exist, a coordinate system requires specification when assessing quantitative change in biodiversity because it is an imprecise term, so there are many different ways in which biodiversity could be measured (Purvis & Hector 2000).

A counterfactual (or indeed any change) cannot be comprehensively specified in the absence of a reference frame: the observer, the parameters they are interested in, and the scale and coordinate system they are using must all be defined first (Supporting Information) even if in practice these are typically implicit or assumed (Bull et al. 2014). The reason for discussing reference frames here is that it is differences in the actors' underlying reference frames that give rise to the family of counterfactuals. To expand, reference frames used for evaluating conservation interventions vary depending on the actor carrying out the evaluation because different actors have different personal values built into their reference frame when constructing a counterfactual (e.g., Bull & Maron 2016). These values may change over time or because actors make different assumptions about the appropriate parameter space and scale for evaluation (e.g., Pauly 1995). Minimizing the sum of perceived differences in an SES, therefore, requires influencing underlying reference frames.

Reference frames can be divergent (e.g., actors using different scales) and even conflicting (e.g., observers making directly contradictory assumptions about value ascribed to biodiversity components) (Hahn et al. 2014). The idea of conflicting reference frames is not new. Schön and Rein (1994) discuss divergent and conflicting reference frames in policy design, respectively, as "disagreements" (resolvable by examining facts) and "controversies" (which tend to be intractable). In conservation, it is well established that focusing on different parameters or coordinate systems can cause conflicting assessments on efficacy (e.g., Naidoo et al. 2008).

Illustration

We considered the preceding concepts relative to examples across different spatial scales. Multiscalar illustration is important because counterfactual evaluation is relevant and necessary for conservation interventions on any spatial scale (Ferraro & Pattanayak 2006). In testing our proposals more thoroughly–and certainly in implementing them–the counterfactuals used would be based on extensive empirical data collected from different actors.

Influence of Aichi Target 11

At the largest scale (international), we considered the global biosphere an SES. Aichi Target 11 is associated with the Convention on Biological Diversity (CBD) (https://www.cbd.int/sp/targets/). This target, set in 2010, was intended to be met by 2020, and there are numerous interested stakeholders seeking to understand the impact of setting this and other Aichi targets (Butchart et al. 2010). Aichi Target 11 states that, "By

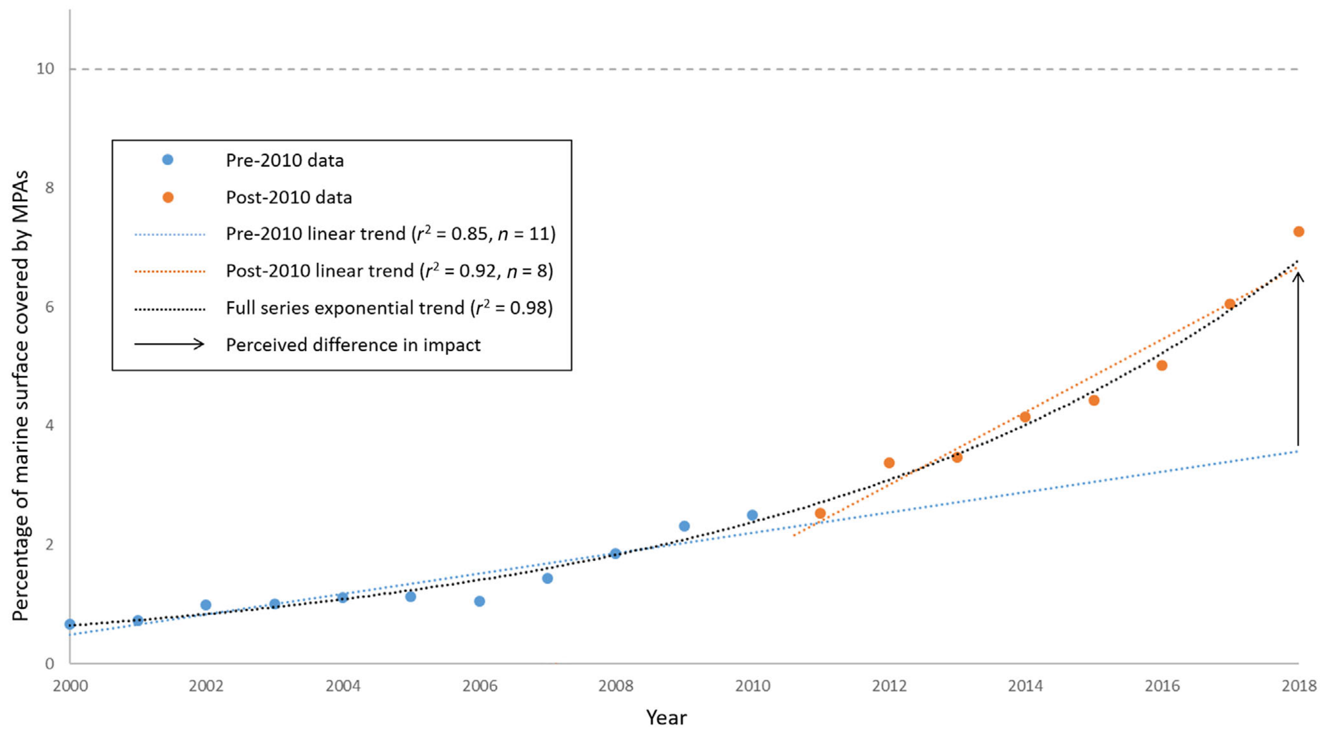

Figure 2. Relationship between global marine protected area (MPA) coverage and time (blue, pre-2010 trend, counterfactual for actor 1; orange, post-2010 trend; black, exponential trend-line fit to entire time series, counterfactual for actor 2; gray, Aichi Target 11, 10% marine surface covered by MPAs). A perceived difference in the impact of setting Aichi Target 11 arises between 2 actors who use different counterfactuals (blue and black lines). Data from UN Environment Programme World Conservation Monitoring Centre (UNEP-WCMC 2019).

2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes." We focused on marine protected areas (MPAs), that are tracked via the World Database on Protected Areas.

Observations seemingly indicate progress toward Target 11, as evidenced by the ongoing trend in MPA coverage (Fig. 2). But the process of setting the Aichi Targets can be viewed as an attempt to stimulate specific actions across the 196 countries that are parties to the CBD. As such, it is reasonable to ask whether there has been any associated change in conservation trends since 2010. Imagine 1 actor characterizes the immediately pre-2010 trend in MPA coverage as linear and takes the post-2010 projection of that linear trend as the counterfactual. Comparing the latter against the observed trend for MPA coverage post-2010, this actor treats the difference between the 2 as resulting from action stimulated by setting Target 11 (Fig. 2). Though not a particularly sound statistical analysis, it represents a plausible interpretation of the data.

A second actor instead argues that a nonlinear trend line fits the 18-year data set better, suggesting the MPA network is on a longer term exponential growth trajectory. This implies that the counterfactual scenario in the absence of the CBD Strategic Plan is also the observed outcome, insinuating (correctly or otherwise) that setting Target 11 had no influence on net MPA outcomes. The perceived difference between these 2 actors regarding the degree to which Target 11 has stimulated additional growth in MPA coverage is clear, despite their using the same biophysical outcome, metric, and data set (Fig. 2).

Australian State Biodiversity Offset Policy

A more pertinent application is in exploring the actions taken toward meeting global conservation policy targets at regional scales, not least because it is on such scales that conservation interventions typically act and ecosystem responses can be monitored over reasonable time scales. Australian state-level biodiversity offset policies require that biodiversity losses via clearance of certain native land cover, as a result of economic development activities, be fully compensated through biodiversity gains on sites with comparable native vegetation elsewhere. These gains can take the form of

Table 1. Perceived difference in outcomes between the regulator (actor 1 [A1]) and researchers (actor 2 [A2], respectively) in the case of 6 different Australian state biodiversity offset policies derived from the formulation of the sum of perceived differences ($\Sigma\Delta_i$) (Supporting Information) and normalized sum of perceived differences.

	A1 assumed counterfactual (% habitat loss per annum)	*A2 assumed counterfactual (proxy, % deforestation per annum)*	*Perceived difference ($\Sigma\Delta_i$)*	*Normalized sum of perceived differences*
New South Wales (NSW) biobanking	0.55	0.21 (min 0.14, max 0.32)	0.34	1.6
NSW major projects	0.35 (min 0.07, max 0.65)	0.22 (min 0.00, max 0.42)	0.13	0.6
Queensland (Qld)	1.00	0.60 (min 0.32, max 1.96)	0.4	0.7
South Australia (SA)	1.48 (min 0.47, max 2.49)	0.09 (min 0.07, max 0.14)	1.39	15.4
Western Australia (WA)	1.50 (min 1.00, max 1.50)	0.19 (min 0.09, max 0.34)	1.31	6.9
Victoria (Vic)	3.12 (min 2.01, max 4.20)	0.45 (min 0.12, max 0.80)	2.67	5.9

Note: Data from Maron et al. (2015).

averted losses (i.e., protection of an area that prevents otherwise near-certain degradation), and as such the counterfactual scenario used to evaluate impacts is critical (Maron et al. 2015).

For 6 offset policies across 5 Australian states, Maron et al. (2015) compared the counterfactual rate of habitat loss assumed by the regulator (the crediting baseline in Fig. 1) against a counterfactual calculated by researchers based on the proxy of recent observed deforestation. These can be considered the counterfactuals assumed by actors 1 and 2, respectively, and clearly the difference is substantial for most states. Using our mathematical formulation (Supporting Information), we calculated the sum of perceived differences between the 2 actors for each state separately (Table 1). We found that even the normalized sum varied by 1–2 orders of magnitude between states; South Australia was a potential outlier (although this may be an artifact of offset data availability for the state [Maron et al. 2015]). In isolation, these figures do not substantially advance understanding of the SES, but if compared with similar statistics calculated for a wide range of regional policy interventions elsewhere (an empirical application of our framework), they could indicate the relative degree of disagreement over policy outcomes.

Species Protection in Uzbekistan

The previous examples had 2 actors, but the framework can be extended to any number and for different biodiversity components. Consider protection efforts on a subnational scale for an IUCN Red List species, the critically endangered saiga antelope (*Saiga tatarica*). In remote northwest Uzbekistan, the Ustyurt saiga population has declined in recent years, and consequently attracted intensive conservation efforts, but the region is also experiencing increasing economic development activity (Bull et al. 2013). New development projects typically seek to mitigate impacts that might exacerbate extinction risk for saigas. So a counterfactual scenario is needed to evaluate the success of mitigation measures relative to the observed trend in saiga population numbers.

We compared 5 possible counterfactual population trends for Uzbek saigas over 15 years against the observed population trend. Three counterfactuals related to known population trends for distinct saiga populations of historically comparable sizes (data from Association for the Conservation of Biodiversity of Kazakhstan), and 2 were hypothetical but realistic counterfactuals (extirpation and population expansion without die offs) (Fig. 3). From initially close alignment, the counterfactuals (which we ascribed to 5 different actors) diverged substantially over time. We tracked this with the annual normalized sum of perceived differences across the family of counterfactuals (Fig. 3) and found dramatic variation in terms of interpreting the impact of protection efforts.

If divergent families of counterfactuals in SES undermine interventions, it is a problem that needs resolution. One solution would be to make the divergence explicit,

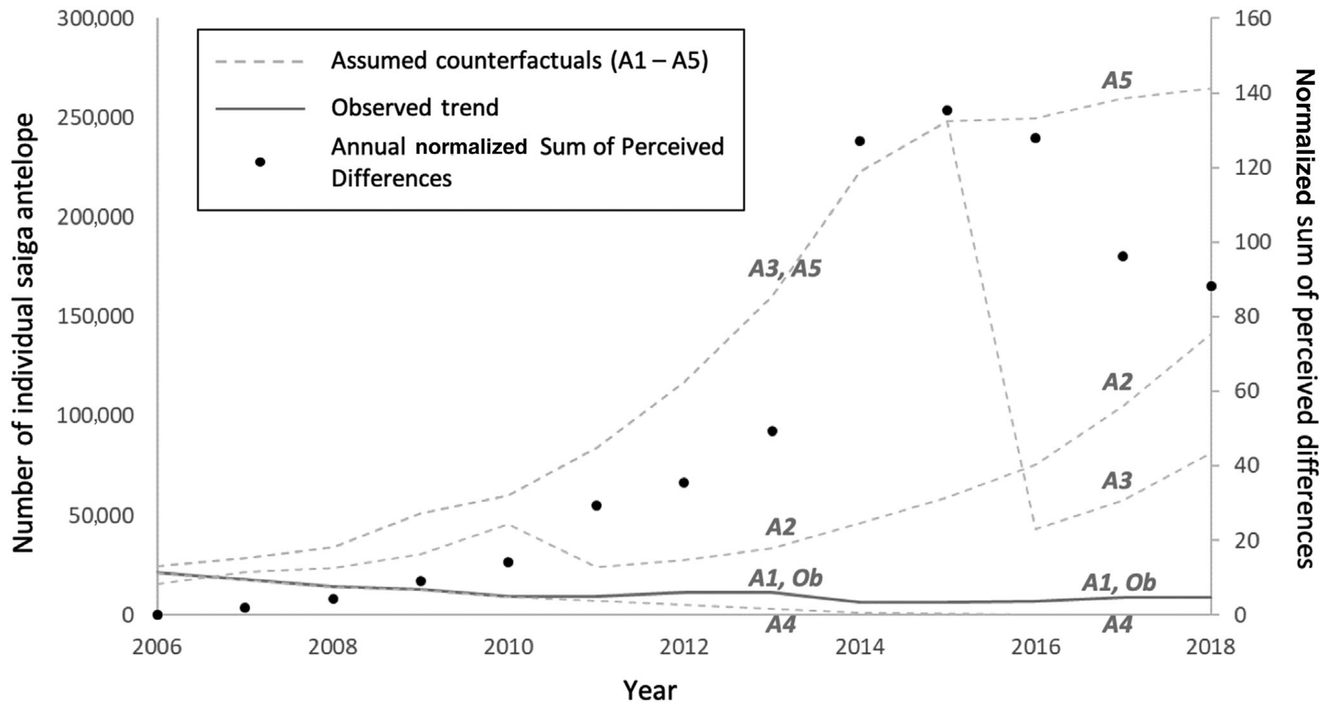

Figure 3. Impact of species protection in northwest Uzbekistan 2006–2018 relative to 5 possible counterfactuals assumed by 5 actors (A1–A5) for the regional saiga antelope (Saiga tatarica) population (primary y*-axis: dashed gray lines, 5 counterfactual population trends; solid line, observed [ob] population trend [A.C.B.K., data]; secondary* y*-axis: block dots, annual normalized sum of perceived differences). All 6 lines labeled, for clarity, at years 2013 and 2017.*

requiring actors to agree to disagree (Biggs et al. 2017). But, alternatively, we considered opportunities to minimize the summed difference across a family of counterfactuals via resolving conflicting frames.

Minimizing Perceived Differences

A starting point is to understand the relevant actors' reference frames. An extensive literature exists on stakeholder analysis in relation to natural resource management (Reed et al. 2009; Cummings et al. 2018). Determining reference frames through stakeholder analysis involves uncovering the perceptions of different actors, associated social discourses, and relationships between actors (Reed et al. 2009; Baynham-Heard et al. 2018). One aspect that needs consideration when implementing our approach is the power dynamics between actors (i.e., differentiation between actors' ability or capacity to influence outcomes). This involves assessing who is represented and who is left out of the decision making and whether the reference frames of some actors are given greater weight (e.g., Smith et al. 2010). This is not the same as weighting for power dynamics when performing our proposed calculations. We avoided the latter because our framework relates specifically to differences in counterfactuals chosen by various actors, rather than actors' ability to act on those differences. Power dynamics are certainly important to our approach if they result in certain stakeholders being completely excluded from consideration, such that their perspective is not incorporated in the sum of perceived differences, a risk that requires careful treatment. Conversely, consideration of the specific nuance of power dynamics–albeit likely an important next step for research on this topic–is more crucial to discussions around steps taken to resolve such differences. Though we begin to explore the latter in this article, the issue of power dynamics deserves further attention in its own right.

Authors have explored options for using understanding of actors' perspectives to design effective conservation (Battista et al. 2018; Cinner 2018; Cummings et al. 2018). We did so based on our formalized structure for reference frames (Supporting Information). So, having isolated different actors' reference frames, we can structure possible approaches to conflict resolution in terms of the 3 key elements (Table 2 & Supporting Information) of reference frames: physical component, social component, and coordinate system.

Table 2. Possible causes of divergence between different reference frames for actors observing conservation interventions.

Component of frame (F)	*Possible areas of divergence*	*Examples of divergence potentially leading to conflict*	*Relevant references*
Physical parameter space (E^n)	number and type of parameters	differences in parameters to incorporate (e.g., objectives and targets of interventions)	Burgman et al. 2011; Maxwell et al. 2015
		differences in physical parameters (e.g., temperature) to include	Poiani et al. 2011
		whether the intervention is treated as dynamic (i.e., incorporation of time as a parameter) or otherwise	Corlett 2016
Observer's personal reference frame (F_{ob})	personal values	differences in perceived relationship between biodiversity and personal well-being	Woodhouse et al. 2015
		whether it is assumed high biodiversity is better than low biodiversity (e.g., species numbers)	Bull & Maron 2016
		differences in intrinsic incentives to engage in conservation interventions	Reddy et al. 2016
	social values	differences among preferences to conserve certain components of biodiversity	Marris 2013
	cultural values	differences in attribution of sacred values versus secular values to components of biodiversity	Daw et al. 2015; Biggs et al. 2017
Coordinate system (C)	choice of indicators	whether measurement of outcomes is in relation to biological diversity or to functionality (and consequent ecosystem service provision)	Naidoo et al. 2008
	choice of scale	whether intervention evaluation is on the spatial scale of individual projects versus a landscape scale	Bull et al. 2014; Maron et al. 2018
		difference in choice of temporal scale over which interventions are evaluated for efficacy	Bull et al. 2014
		whether historical context is included	Willis & Birks 2006

Conflict in the Physical Component

The physical component of the reference frame (i.e., measurable biophysical quantities, whether biotic or abiotic [Supporting Information]) is inevitably associated with development of conservation objectives. Consequently, resolution of conflict in the physical component of the reference frame relates strongly to conservation objectives and targets.

Structured decision making (SDM), an approach designed to "systematically incorporate participant values, objectives and knowledge in decision-making" [Addison et al. 2013]), can help one develop objectives for natural resource use, given competing actor values (e.g., Robinson et al. 2016). Further, despite involving explicit expression of conflicting reference frames, SDM can strengthen consensus among groups during decision making (Priem et al. 1995), such as design and evaluation of interventions. Indeed, experts typically perform better at making decisions based on well-structured discussions with peers rather than alone (Burgman et al. 2011). Incorporating monitoring and evaluation into SDM is well established (Lyons et al. 2008). Considering these factors, SDM provides a practical platform for using group consensus to reduce conflicts between stakeholders with different reference frames.

When there are intractable differences between stakeholders, consensus may be impossible. International policy decisions can be seen as negotiated decisions constructed through multiactor interactions (Daniels et al. 2012). In such cases, for example, the development of quantifiable environmental targets, Maxwell et al. (2015) propose allowing room to maneuver, promoting ambiguity in targets with the goal of building trust, cooperation, and consensus. This is akin to allowing ambiguity in specification of the parameter space for the physical component of conflicting frames to promote convergence in social components of those reference frames. An obvious disadvantage is that ambiguity in the definition of any component of the reference frame makes rigorous evaluation impossible–so the approach sacrifices short-term measurability to enable long-term resolution.

Conflict in the Social Component

More diverse groups often frame likely outcomes more accurately (Sutherland & Burgman 2015). But this is not only relevant to experts and decision makers. Addison et al. (2013) suggest using participatory SDM to increase broader social acceptance of conservation objectives. That is, through participatory exercises featuring diverse nonexperts, interventions can be designed that integrate multiple social reference frames. Such reference frames typically involve several layers of complexity: deeply held values, important interests, local knowledge, and socially embedded conflicts among them (Daniels et al. 2012). Similarly, participatory modeling is designed to integrate a diversity of perspectives (i.e., social reference frames) through, for example, use of role-playing games (Jones et al. 2007) for both specialist and nonspecialist participants. In seeking consensus across heterogeneous groups of actors, there are many formal processes available (e.g., stakeholder engagement, scenario planning [Cummings et al. 2018], and workshops to facilitate alignment between individual social values [Kenter et al. 2015]).

Beyond participatory mechanisms, an extensive literature is concerned with the degree to which personal reference frames can be modified, including for conservation (Iftekahr & Pannell 2015). Consider anchoring when estimating numerical outcomes; individuals generally make minor adjustments to some initial value to come to their answers. The initial (anchoring) value is linked to the individual's personal cognitive frame and influenced by the formulation of the question itself (Tversky & Kahneman 1974). Consequently, the extensive theoretical basis underlying the construction of personal reference frames could be leveraged to support resolution of conflicting reference frames, for instance, by purposefully creating common, appropriate anchoring points for evaluating conservation interventions by a range of actors (e.g., the quantitative extent of remaining habitat evaluated as acceptable [Cinner 2018]).

The power of setting expectations for assessment is also relevant in terms of qualitative presentation of interventions; policies are likely deemed more acceptable when presented without loaded terms. Again, the presentation of interventions can thus be used proactively to help set personal reference frames among actors and reduce potential conflicts, albeit with associated ethical considerations (e.g., Rothschild 2000). Alternatively, personal reference frames may be open to modification through management of appropriate incentives. Although, prosaically, these could be the provision of financial incentives, they could also be intrinsic incentives related to personal desires or values. Some stakeholders, for example, may be better motivated by attachment to the land than by financial incentives (Reddy et al. 2016).

Of course, substantial components of personal reference frames are deeply held and cannot be modified. Individuals distinguish between sacred and secular values (Tetlock et al. 2000). This forms part of an actor's personal reference frame, and they may consider it unacceptable for a conservation intervention to exchange a sacred value for a secular one (Daw et al. 2015). In that case, the solutions would be to explicitly identify sacred values held by different actors and ensure that they are not jeopardized by interventions. Such an approach–requiring an iterative process through which trust and cooperation is built between key parties on a seemingly intractable issue–involves negotiating shared belief structures (Cummings et al. 2018) (e.g., squaring different "mental models" held by stakeholders [Biggs et al. 2017]). Alternatively, the intervention could be redesigned around secular values, for which examples in the literature go back decades (Cummings et al. 2018).

Conflict in Coordinate Systems

One solution to conflict caused by using different coordinate systems is accepting the need to track multiple indicators (e.g., Butchart et al. 2010). Approaches that combine multiple indicators are used widely elsewhere (e.g., finance; Engle & Gallo 2006). Where this is not satisfactory–say, due to the increased resource requirements–an alternative is again participatory approaches that facilitate consensus between actors on coordinate systems rather than parameter spaces.

Conflict arising from differences in the spatial or temporal scale for evaluation may arise because scales are implicitly assumed by different actors (Bull et al. 2014). An explicit statement of the scale, as part of counterfactual construction, is therefore 1 straightforward approach toward avoiding conflict. A more nuanced approach would be to consider the appropriate counterfactual scenario for different spatial scales at a given moment. Maron et al. (2018) recommend that the scope of the reference frame be explicitly expanded to approach the largest possible overarching frame. Doing so is comparable with other calls in the literature to evaluate conservation interventions with very long-term time scales (e.g., Willis & Birks 2006). Finally, Pearson (2016) suggests that actors construct their personal reference frames for conservation based partly on the spatial scale in question. Setting the scale could itself be a means for influencing personal reference frames.

Conclusions

All biodiversity conservation interventions take place in an SES, and if an SES contains more than 1 actor, this may give rise to a family of counterfactuals. The use of a family of different counterfactuals by different actors will cause perceived differences in the impacts of (change attributable to) an intervention, even when all actors agree on outcomes. This may lead to disagreement among actors on the efficacy of interventions and undermine efforts to conserve biodiversity. But, because no counterfactual can be considered definitive, multiple different impact evaluations can be performed or proposed by different actors, more than 1 of which may be considered valid. Therefore, we developed the basis for evaluating a sum of perceived differences between actors (defined mathematically in Supporting Information) and explored approaches for minimizing perceived differences. An important next step would be to test the conceptual framework we developed against extensive empirical data in terms of the value of our proposed sum in different SES and associated implications and in terms of finding the limits to which counterfactuals may be considered valid. Our work provides an exploratory theoretical basis for better quantifying, understanding, and ultimately managing multiple diverse perspectives on nature conservation in SES.

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Supporting Information

A mathematical formulation for the sum of perceived differences (Appendix S1), a formal definition of a reference frame (Appendix S2), and a brief review of the use of the term *reference frame* in a variety of scientific disciplines (Appendix S3) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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