CONTRIBUTED PAPER

The relative conservation impact of strategies that prioritize biodiversity representation, threats, and protection costs

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Abstract

Despite exponential increases in the coverage of protected areas (PAs) over recent decades, global biodiversity continues to decline. One explanation for this lack of success is that the efficacy of conservation prioritization strategies is rarely measured in terms of conservation "impact," which requires comparing proposed PA networks to a counterfactual scenario in which no intervention is applied. This approach contrasts with measuring efficacy using surrogates for conservation impact, such as the extent, total biodiversity value, or representativeness of a proposed PA network. However, implementing an experimental counterfactual scenario is difficult because of time, funding, and ethical constraints. Here, we use an alternative and complementary approach: an *ex-post* analysis with counterfactual outcomes measured using historical empirical data on changes in biodiversity in unprotected landscapes. This approach allows for the comparison of different retrospectively implemented prioritization strategies to a real counterfactual outcome. In our analysis, we predict the impact of several alternative PA prioritization strategies in Queensland, Australia, using high-resolution datasets of vegetation clearing, habitat type, and land acquisition cost. Our results show that achieving conventional conservation targets does not equate to achieving impact, and that alternative, and relatively simple, prioritization strategies can achieve far greater impacts.

KEYWORDS

conservation costs, conservation impact, counterfactual, protected areas, Queensland, representation, spatial prioritization, systematic conservation planning, threats, vegetation clearing

1 | INTRODUCTION

Despite increasing conservation efforts worldwide, evidence of continued declines in biodiversity (Butchart et al., 2010; Hoffmann et al., 2010; Tittensor et al., 2014) has called into question the efficacy of current conservation prioritization methods (Carwardine, Klein, Wilson, Pressey, & Possingham, 2009; Pressey, Weeks, & Gurney, 2017). Since the inception of systematic conservation planning (Margules & Pressey, 2000; Moilanen, Wilson, & Possingham, 2009; Pressey, 2002), a myriad of spatial prioritization methods have been developed and

implemented, each with the ultimate goal of maximizing the persistence of biodiversity using limited conservation funds. However, analyses of protected area location have shown that even systematic approaches are susceptible to "residual" biases, whereby areas of high latitude, poor soil quality, and low economic value receive disproportionately high levels of protection, while high-quality areas particularly susceptible to exploitation are under-represented (Devillers et al., 2015; Joppa & Pfaff, 2009).

The emergence of residual biases in systematic approaches has been largely attributed to two factors: the failure to frame conservation goals and objectives in terms of impact (Pressey et al., 2017; Pressey, Visconti, & Ferraro, 2015), and the difficulty of empirically measuring conservation impact as a guide to setting priorities for conservation planning (Bottrill & Pressey, 2012; Ferraro & Pattanayak, 2006; Ferraro & Pressey, 2015; McIntosh et al., 2018; McIntosh, Pressey, Lloyd, Smith, & Grenyer, 2017). Conservation impact can be measured only by comparing outcomes from an intervention to outcomes from no intervention (referred to as "counterfactual" outcomes in the conservation literature, sensu Ferraro, 2009). However, the rigorous experimental procedures standard in other scientific fields, involving control (i.e., counterfactual) and treatment groups, are impractical in conservation science, because they would involve implementing multiple alternative conservation prioritization strategies in many replicate regions over periods of time relevant to conservation (i.e., several decades). It would also be an ethically questionable procedure, because counterfactual planning regions would receive no conservation interventions (or interventions that were known to be suboptimal) when they might be urgently needed.

A variety of tools are employed by conservation practitioners, including the implementation of protected areas (PAs), regulation of threats to biodiversity (e.g., land clearing restrictions), and management of biodiversity (e.g., invasive species control), among others. In this article, we focus on the use of PAs, which are one of the most widely adopted tools for use in systematic conservation planning (Margules & Pressey, 2000). Today, the predominant approach to conservation planning involves designing a network of complementary and representative PAs, which typically involves setting a specific target (e.g., total or proportional area) for each biodiversity feature of interest within the planning region (Kukkala & Moilanen, 2013). Other methods focus on designing PA networks that also, or alternatively, focus on other attributes, such as maximizing connectivity between PAs (Beger et al., 2010), or minimizing costs of protection (Naidoo et al., 2006). Representation targets are widespread in conservation policy and practice, often serving as the primary objective of national and multinational reserve systems supported by

millions of dollars of public and private conservation funding (e.g., Commonwealth of Australia, 2005; Fernandes et al., 2005; UNEP-WCMC, 2008). However, many of these approaches disregard an essential component of conservation impact: threats to biodiversity. Threats can be difficult to incorporate into conservation planning because detailed spatial datasets are not always available, and there is considerable uncertainty associated with using historical data on threats (e.g., vegetation clearing, pollution, fishing pressure) to predict spatio-temporal patterns of threats in the future. Consequently, there is a large gap in the conservation science literature concerning the relative importance of various targets, such as biodiversity representation, threat mitigation, and cost minimization, for maximizing conservation impact (Pressey et al., 2017).

There are three methods to overcome the problem of identifying counterfactual conditions to estimate the conservation impact of alternative prioritization strategies. The first method involves sophisticated quasiexperimental matching techniques, whereby existing PAs are matched to unprotected areas with similar biophysical and socioeconomic characteristics to correct for the non-random allocation of PAs (Ahmadia et al., 2015; Andam, Ferraro, Pfaff, Sanchez-Azofeifa, & Robalino, 2008; Jones & Lewis, 2015; Joppa & Pfaff, 2011). These matched unprotected areas serve as a pseudo-counterfactual to which the outcomes of PAs can be compared. However, matching techniques have limited use in conservation planning, because they can be used only to assess the particular prioritization strategy used to implement an existing PA network, and not to compare and estimate the impacts of a range of different strategies. The second method is to compare alternative prioritization strategies using *ex-ante* modeling of future landscapes to predict counterfactual outcomes (Monteiro et al., 2020; Newburn, Reed, Berck, & Merenlender, 2005). These are particularly useful for identifying areas for potential protection. However, predicting impacts with an *ex-ante* approach relies upon a range of assumptions and uncertainties about spatial and temporal changes in threats and biodiversity in the absence of, and in response to, protection.

In this article, we use a third, *ex-post*, approach that is underutilized in the conservation planning literature, and overcomes some of the shortfalls of the approaches listed above. The *ex-post* approach involves measuring impacts retrospectively, which is particularly useful because it provides a real, empirical, and counterfactual scenario. Biodiversity outcomes are then estimated under alternative protection scenarios and compared to this observed counterfactual. For our analysis, we use empirical data on historical changes in vegetation cover across a range of vegetation types in Queensland, Australia, between 2006 and 2016. We compare how biodiversity outcomes differ when four alternative conservation prioritization strategies

are implemented: (a) prioritizing low-cost areas for protection, (b) maximizing the representation of biodiversity features in PAs, (c) prioritizing areas facing high threat from land clearing for protection, and (d) maximizing the representation of biodiversity features within PAs while also prioritizing areas facing high threat from land clearing. We also use high-resolution datasets of land valuation to explore the cost-efficiency of each strategy, and to explore how impacts vary according to available budgets.

2 | METHODS

2.1 | Case study

For our analysis, we used the case study of Queensland, Australia. Queensland provides a useful test case because it is a large state (185 million ha) containing a broad range of vegetation types, from semiarid woodlands to tropical rainforests. Queensland has experienced extensive and rapid vegetation loss since European settlement. The cover of native forests, shrublands, and heathlands in Queensland at the time of European settlement is estimated to have been approximately 80%, but has since been reduced to approximately 30%, primarily for the creation of cattle grazing lands (Bradshaw, 2012; Evans, 2016). Spatial conservation prioritization is therefore both urgently needed and highly consequential in Queensland.

2.2 | Planning units and loss of woody vegetation

The planning units for our analysis consisted of land property parcels in Queensland. Land parcels in Queensland are variable in size and irregularly shaped. The size distribution of parcels in Queensland is highly positively skewed, with a median area of ~0.1 ha, and 82% of parcels covering less than 1 ha. The majority of small parcels are used for housing in urban areas, while larger parcels are predominantly used for agriculture in more remote rural areas. The analysis was restricted to parcels that were outside present-day PAs, to ensure that we could obtain a reliable counterfactual measure of land clearing in the absence of protection. All parcels within 1 km of PAs were also removed from the analysis to avoid potential confounding differences in vegetation clearing patterns in areas proximal to PAs.

The conservation goal was to minimize the loss of vegetation across 29 broad vegetation groups (Neldner, Niehus, Wilson, McDonald, & Ford, 2014), using the available budget. We assumed that all parcels were available for purchase. We tested how well each prioritization

strategy could achieve the conservation goal over a period of 10 years, from 2006 to 2016. Each prioritization strategy could protect a set of parcels in 2006, after which we assumed that protected parcels would lose no vegetation (but see below for consideration of displacement of land clearing). For each strategy, the entire budget had to be spent in 2006, and no further protection was allowed during the study period from 2006 to 2016.

We measured the realized loss of woody vegetation on unprotected parcels using the Statewide Landcover and Trees Study (SLATS), derived from Landsat satellite imagery and field surveys to measure woody vegetation clearing across Queensland. To estimate the extent of woody vegetation in 2006 and 2016, we combined SLATS data with data on the extent of woody vegetation in 2016 (Queensland Department of Environment and Science, 2018; Queensland Government, 2018). We then created a layer of woody vegetation in 2006 under the assumption that all woody vegetation present in 2016 and registered as cleared between 2006 and 2016 was present in 2006.

2.3 | Budget constraints and analyses

To estimate the conservation acquisition cost of each parcel, we used statutory unimproved land valuations by the Queensland Valuer-General for all rateable land parcels (Queensland Government, 2008). This dataset included valuations between 2002 and 2006. To account for inflation, we standardized these land valuations to Australian dollars in 2006 (2006 AUD) using the average annual Australian consumer price index (Australian Bureau of Statistics, 2017).

In the primary analysis, we set the total budget to 1 billion 2006 AUD to purchase land in 2006 for protection over the entire period from 2006 to 2016. This is equivalent to 100 million 2006 AUD per year, which is within the range of annual expenditures by Queensland's Environmental Protection Agency in 2006 (Queensland Government, 2006). However, we also tested how the impact of each strategy varied according to the available budget by varying the total budget from 200 million to 10 billion 2006 AUD, with increments of 200 million AUD. Details of alternative budget analyses are provided in Supporting Information.

2.4 | Prioritization strategies and the counterfactual

We measured the impact of all prioritization strategies relative to a counterfactual scenario in which no parcels were protected over the period of analysis. All prioritization strategies were designed with the software Marxan (Watts

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et al., 2009). All strategies attempted to achieve their respective objectives using only the specified budget (see section below). In Marxan, this was implemented by setting a cost threshold that could not be exceeded. For each strategy, we performed the Marxan run with 1,000 iterations, and thereafter took the best solution from each run. This method was also used for the budget analysis, whereby we performed 100 iterations of each strategy for each budget interval (200 million, 400 million, and so on).

We compared four different strategies. First, the cost-only strategy, which effectively prioritized parcels with the lowest cost per unit area. This strategy was implemented by treating all parcels as a single biodiversity feature, and setting protection objectives to 100%. This strategy, therefore, simply maximized the total amount of area protected using the available budget. Second, the threat strategy prioritized parcels that were expected to face high levels of threat. We assigned each parcel a threat score, measured as the extent of land clearing in the 10 years prior to the beginning of the planning period (1995–2005) within 20 km of the parcel's centroid. This score was then multiplied by the parcel area. This threat score, therefore, assigned conservation value to each parcel based on the amount of land under threat (parcel size) and the intensity of threats in the area. We implemented this strategy by setting threats as a single feature to be maximized (objective of 100%) using the available budget. Third, the representation strategy attempted to represent 30% of each of 29 woody broad vegetation groups within Queensland (Neldner et al., 2014). We also tested alternative representation objectives (50 and 90%), the results of which are available in Supporting Information. Fourth, the representation and threat strategy attempted to represent all broad vegetation groups while also prioritizing areas under high levels of threat from land clearing. For this strategy, we used the same representation objectives as those from the representation-only strategy (including supporting analyses with objectives of 50 and 90%), and the same threat objectives as those from the threat-only strategy. Then, in Marxan, we set the vegetation groups and the threat feature as two distinct feature types with equal priority. Full details of the prioritization methods are provided in Supporting Information.

2.5 | Measures of impact

We compared each prioritization strategy using three different impact metrics, all relative to the counterfactual scenario. The first metric was the total area of mitigated vegetation loss within each of the 29 woody broad vegetation groups present in Queensland (Neldner et al., 2014).

Because some broad vegetation groups naturally cover large extents while others are restricted, our second metric was the area of mitigated vegetation loss in each broad vegetation group in proportion to its total extent in 2006. Our third metric was a relative impact score that weighted more heavily the preservation of broad vegetation groups according to their rarity (i.e., higher weighting to groups with smaller extents in 2006) and their historical rates of clearing (i.e., higher weighting to groups that had a lower proportion of their pre-European extent remaining in 2006). Full details and sensitivity analyses for this metric are provided in Supporting Information.

We also measured how evenly impact was distributed across groups. We refer to this as "impact equality," based on the protection equality metric of Chauvenet, Kuempel, McGowan, Beger, and Possingham (2017). Impact equality was higher if the proportion of avoided loss was more evenly distributed across groups. Notably though, impact equality alone is not a good measure of impact, because it does not consider the total extent of avoided loss. For example, a strategy that saved an equally small proportion of each group would have higher impact equality than a strategy that saved higher, but more variable, proportions of broad vegetation groups. Full details and formulae for this metric are available in Supporting Information.

2.6 | Displacement of land clearing

One of the criticisms of PAs is that their positive effects can be offset by "displacement," also known as "leakage." Displacement occurs when, after protection, threatening processes shift to nearby unprotected areas (Ewers & Rodrigues, 2008; Moilanen & Laitila, 2016; Renwick, Bode, & Venter, 2015). To account for the possibility of displacement after protection, we created a spatial displacement model. In the displacement model, once a parcel was protected by a prioritization strategy, all land clearing that would have occurred in that parcel between 2006 and 2016 was distributed to unprotected parcels within a 5 km radius. We also investigated alternative displacement distances (1, 10, and 20 km) and report on these in Supporting Information. The spatial model was employed in ArcGIS 10.4.1 using custom python code (available upon request).

3 | RESULTS

3.1 | Counterfactual outcomes

The final analysis included 126,232 land parcels, covering 34,996,900 ha in Queensland, across which there were

19,442,148 ha of remnant woody vegetation in 2006. Between 2006 and 2016, in the counterfactual scenario (i.e., in reality), these parcels lost 1,014,118 ha of woody

vegetation. Vegetation loss was uneven across broad vegetation groups, both in terms of area and as a proportion of their extents in 2006 (Table S1). The most extensively

TABLE 1 Matrix of spatial overlaps between strategies in the primary analysis with a budget of 1 billion 2006 AUD for the 10-year period from 2006 to 2016

Hectares				
	Cost	**Threat**	**Representation**	**Representation/threat**
Cost	**19,380,662**			
Threat	11,905,338	**13,411,204**		
Representation	9,547,457	6,230,764	**11,015,431**	
Representation/threat	8,753,898	4,846,359	6,738,626	**9,721,435**
Percentage				
	Cost	**Threat**	**Representation**	**Representation/threat**
Cost	100%			
Threat	57%	100%		
Representation	46%	34%	100%	
Representation/threat	43%	27%	48%	100%

Note: The diagonal in the first matrix represents the total hectares protected with each strategy (bold text).
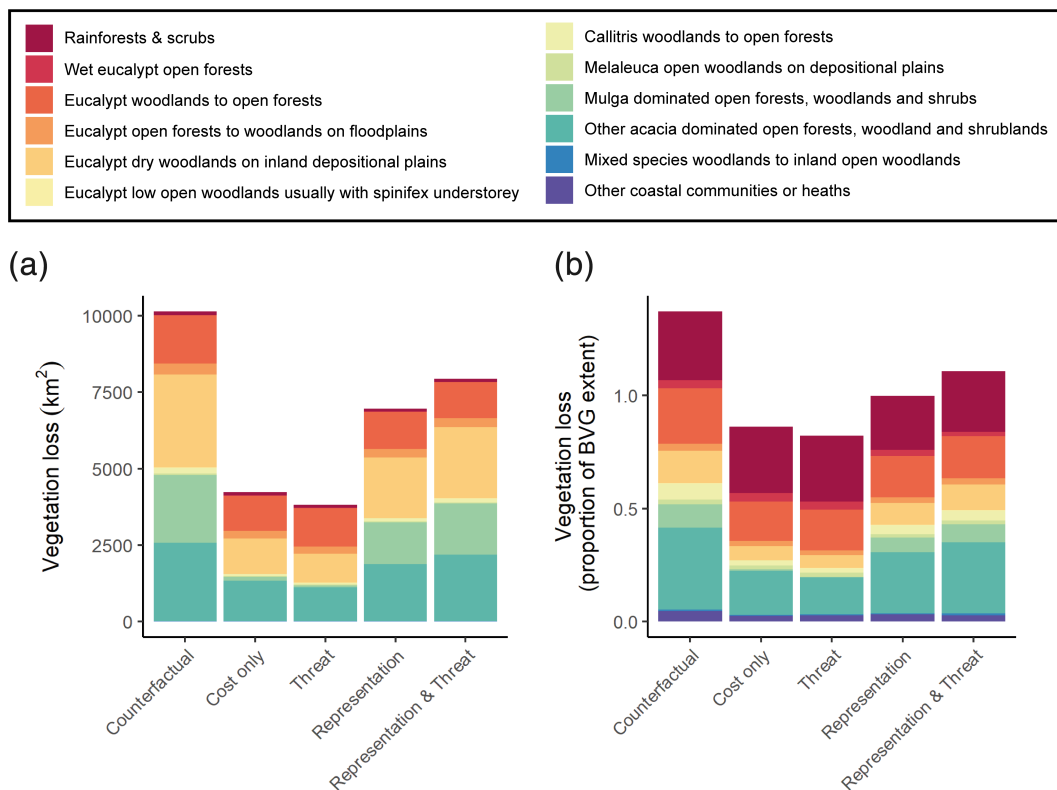

FIGURE 1 The impact of each strategy relative to the counterfactual scenario. Panel (a) shows impact measured as the total area of mitigated woody vegetation loss within each broad vegetation group (BVG). Panel (b) shows impact with mitigated vegetation loss measured in proportion to the extent of each broad vegetation group in 2006, such that a score of 1.0 means that all vegetation loss was mitigated. Note that broad vegetation groups have been simplified into the above 12 categories (according Neldner et al., 2014) for ease of interpretation (see Table S1 for details of classification). For the primary analysis, all impact metrics considered the 29 woody broad vegetation groups present in Queensland. For this analysis, the budget was set to 1 billion 2006 AUD for the 10-year planning period

cleared broad vegetation group in terms of area was dry woodlands dominated by *Eucalyptus populnea* (poplar box) or *E. melanophloia* (silver-leaved ironbark), losing 267,895 ha within analyzed parcels. *Acacia harpophylla* (brigalow) dominated open forests and woodlands was proportionally the most extensively cleared broad vegetation group, losing 27% of its extent between 2006 and 2016.

3.2 | Impacts of prioritization strategies

In the primary analysis (budget of 1 billion 2006 AUD), the total area (and number of parcels) protected were as follows: cost strategy—19.4 million ha (6,080 parcels); threat strategy—13.4 million ha (5,605 parcels); representation strategy—11.0 million ha (7,943 parcels); representation/threat strategy—9.7 million ha (7,748 parcels). Spatial overlap between strategies varied from 27% (between the threat and representation/threat strategies) to 57% (between the cost and threat strategies; Table 1).

When measuring impact in terms of the total area of avoided loss, a threat prioritization strategy was most effective (Figures 1a and 2a). A threat prioritization strategy prevented the loss of 633,712 ha of vegetation (380,406 ha not prevented), while a cost prioritization strategy prevented the loss of 586,670 ha (427,448 ha not prevented), a representation strategy prevented the loss of 223,620 ha (790,498 ha not prevented), and a representation/threat strategy prevented the loss of 218,371 ha (795,747 ha not prevented; Table S2). A threat prioritization strategy was also most effective when measuring impact proportional to the extent of each broad vegetation group (Figures 1b and 2b), and when rare and historically cleared broad vegetation groups were weighted more heavily (Figure 2c). These results were consistent, regardless of representation objectives (Figure S3) and when displacement of land clearing was considered (Figure S4).

For all three metrics of impact, changing the budget did not affect which strategy was most effective (Figures 2a–c). Notably though, the relative difference in impact between strategies increased as the budget
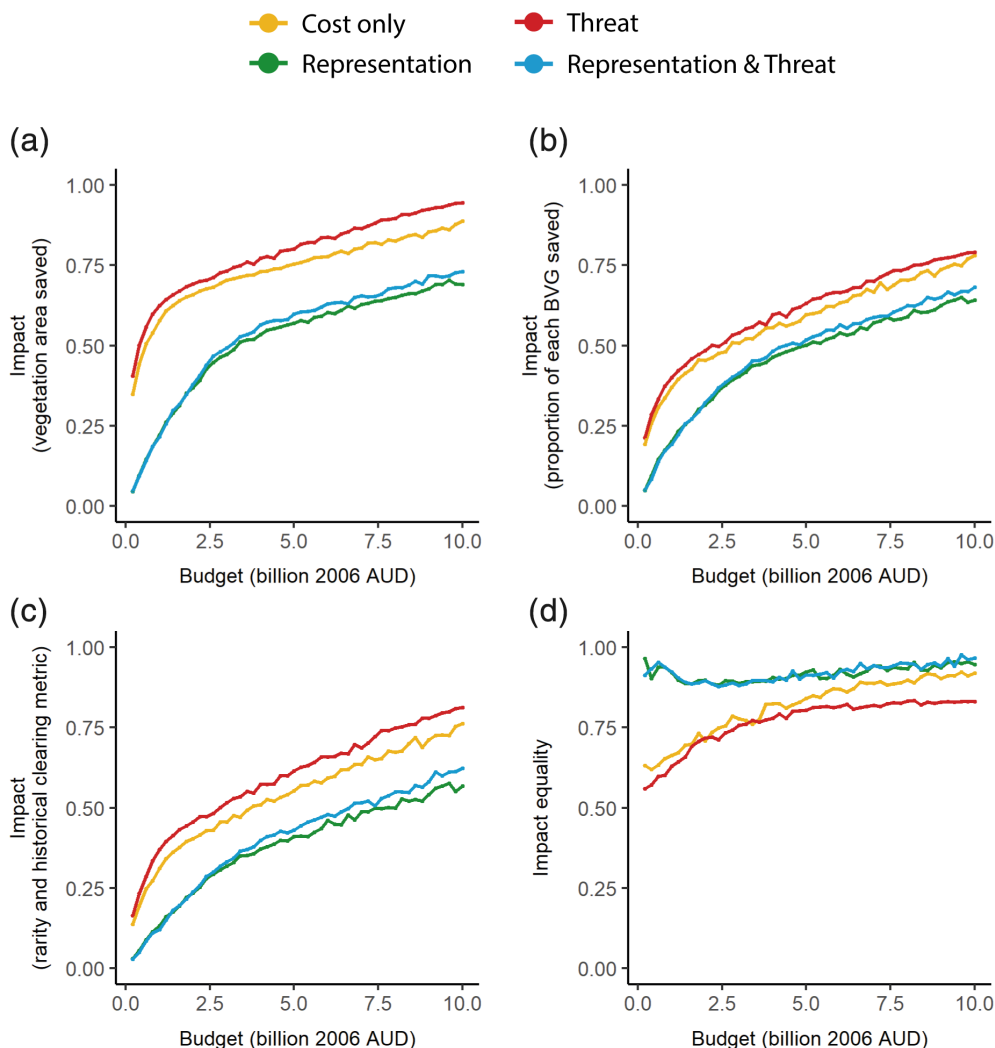

FIGURE 2 The effect of budget on the impact of alternative prioritization strategies according to different metrics. Panel (a) shows the impact of each strategy, measured as the total area of mitigated woody vegetation loss relative to the counterfactual scenario, such that a score of 1.0 means that all vegetation loss was mitigated. Panel (b) shows impact with mitigated vegetation loss measured as the sum of the proportions of mitigated vegetation loss in each broad vegetation group (BVG) relative to the counterfactual. Panel (c) shows impacts with proportions from panel (b) weighted according to the rarity of each broad vegetation group and how extensively it was cleared prior to 2006. Panel (d) shows the impact equality of each prioritization strategy, based on the "protection equality" metric from Chauvenet et al. (2017). Full details and formulae for each metric are provided in Supporting Information

increased. The impact of all strategies would presumably converge at very large budgets (over 10 billion AUD), when all strategies would by necessity protect the same parcels. There were also diminishing returns on conservation investment for all four prioritization strategies (Figures 2a–c). Although cost and threat strategies had higher overall impacts, representation and representation/threat strategies had higher impact equality, particularly at lower budgets, but this difference narrowed as the budget increased (Figure 2d).

4 | DISCUSSION

There is an alarming lack of empirical analyses estimating the impact of modern approaches to conservation priority setting (Ferraro & Pattanayak, 2006; Pressey et al., 2017). Instead, much of the science and practice of conservation prioritization has focused on developing plans that efficiently achieve specific targets (IUCN-WCPA, 2008; National Reserve System Task Group, 2009), while the impacts of achieving such targets are largely unknown. We offer an empirical *ex-post* approach that allows impacts to be estimated by comparing estimated outcomes to a real counterfactual scenario. This approach allows comparison of any number of hypothetical PA systems, rather than being restricted to measuring the impact of existing PA networks, as is the case with matching analyses (Ahmadia et al., 2015; Andam et al., 2008; Jones & Lewis, 2015; Joppa & Pfaff, 2011). Importantly, our results illustrate that reaching any specific target (e.g., cost minimization, threat prioritization, or biodiversity representation) does not guarantee that such an approach will lead to benefits in terms of impact, and maximize return on investment. Thus, it is essential that conservation planners use counterfactual-based measures of conservation impact to assess the relative efficacy of any proposed strategies.

Our results offer empirical support for the argument that equal-proportion representation objectives could have suboptimal return on investment when measuring benefits in terms of conservation impact. Interestingly, the strategy that incorporated representation, threat, and cost objectives also performed poorly. This is a counterintuitive result because it would be expected that considering all of these components would be a good approach to maximizing impact. However, in Queensland, loss of woody vegetation was unequal across vegetation types (Table S1) over the study period. This suggests that the objectives of representation and threat mitigation were antagonistic. When representation was included as the only objective, the strategy attempted to ensure that all vegetation types—including those unlikely to be cleared—were protected in equal proportions. In

contrast, when threat objectives were included without representation goals, the strategy prioritized the most threatened vegetation types, and left less threatened types unprotected. In terms of impact, leaving less threatened vegetation types unprotected was effective because they were not cleared throughout the study period. Including both threat and representation objectives (i.e., the representation and threat prioritization strategy) was ineffective because it was either impossible or very costly to protect the most threatened locations, while also ensuring representative protection across all vegetation types.

In contrast to the representation-based approaches, cost-minimization and threat prioritization were significantly more cost-effective at mitigating vegetation loss across vegetation types (Figures 1 and 2). For example, the budget required by the threat prioritization strategy to prevent 50% of the vegetation loss that occurred in the counterfactual scenario was only ~600 million 2006 AUD, while the budget required to achieve the same impact using the representation strategy was ~3.4 billion 2006 AUD (Figure 2). Although both the cost and threat strategy achieved similarly high impacts, the partial spatial overlap between the two strategies (57%) indicates that impacts were achieved in somewhat different locations, through different means. Cost-minimization simply protected a large amount of land (Table 1), and mitigated a large amount of vegetation loss in doing so. Because in Queensland there are a large number of low-cost, high-threat locations (Sacre, Pressey, & Bode, 2019), many high-threat locations were inadvertently protected. Threat prioritization, on the other hand, protected less land, but was more effective at targeting land at risk of imminent vegetation loss. Notably, however, protecting a large amount of land, as in the cost strategy, might become less efficient when considering other conservation costs, such as management costs, which are likely to be higher for larger PA networks. Other costs, such as implementation costs (e.g., bureaucratic processes) and transaction costs (e.g., fees for lease processing and negotiation), are also likely to be higher for strategies that require the purchase of a greater number of parcels. Our results indicate that such costs might be lower for cost and threat prioritization strategies, which required the purchase of fewer parcels than representation-based strategies.

Our observation that threat prioritization is an effective way to achieve high impact aligns with those of other analyses that utilize an *ex-ante* model-based approach to generate counterfactuals. Visconti, Pressey, Segan, and Wintle (2010), for example, found that a strategy that attempted to prioritize sites most likely to lose biodiversity (i.e., threat prioritization) was generally more effective than prioritizing sites that would contribute to maximizing biodiversity within the PA network.

Similarly, Monteiro et al. (2020), found that prioritizing sites most likely to lose vegetation, from both habitat clearing and climate change, outperformed, in terms of impact, a strategy that attempted to represent all biodiversity features.

Importantly, we wish to emphasize that these results support the conclusion that conservation strategies should aim to achieve representative impacts (i.e., loss prevented across a representative sample of biodiversity), rather than representative protection, because representative protection might not necessarily lead to representative impact in planning regions where biodiversity loss is unequal across biodiversity types (although it might be effective where biodiversity loss is more homogeneous across types). Although we assess only the impact of a fairly rudimentary approach to representation, these results warrant some concern with respect to conservation policy and practice. In Queensland, for example, the protected area strategy places a great degree of importance on representative protection and associated objectives, such as comprehensiveness and adequacy (Fernandes et al., 2005; Queensland Government, 2017). Such approaches are dominant not only in Queensland, but also on national and international scales (Commonwealth of Australia, 2005; UNEP-WCMC, 2008). Certainly, the use of these systematic methods is a great step forward in the right direction towards achieving better conservation outcomes. However, of paramount importance is testing these prioritization strategies using counterfactual-based measurements of impact, and adjusting conservation practice according to the resulting evidence base.

There are several limitations to this analysis that must be considered. First, our _ex-post_ method assumes that areas selected for protection would not have lost any vegetation. While this might be valid for outright purchase in Queensland, in other cases, biodiversity loss could still occur because of noncompliance with formal protection and limitations in management effectiveness (Coad et al., 2019; Geldmann, Manica, Burgess, Coad, & Balmford, 2019). Furthermore, protection alone might do little to abate other threats to biodiversity, such as natural disasters, pollution, and invasive species (Allek et al., 2018); future analyses should focus on incorporating these factors. Second, an _ex-post_ method measures impacts only retrospectively, and the estimated impacts from this method might not apply into the future. However, our general observation that threat prioritization is more effective than a representation or biodiversity-focused approach is consistent with _ex-ante_ predictive methods that estimated future impacts (Monteiro et al., 2020; Visconti et al., 2010). Finally, this analysis considers only the case study of Queensland. Because the impact of any given strategy depends highly upon the spatial distribution of threats, costs, and biodiversity (Sacre, Bode, Weeks, & Pressey, 2019), results might differ in other planning regions, and when using alternative measures for these factors. For example, impacts might differ substantially in marine planning regions, because the costs associated with marine conservation are typically opportunity costs (forgone economic profits) and management costs rather than acquisition costs (Hunt, 2013). Similarly, impacts might differ when using other measures of biodiversity, such as species richness or functional diversity, rather than vegetation types. A key knowledge gap to be explored in further analyses is how the relative impact of strategies might change in response to the spatial relationship between these factors, and particularly how results might differ in regions where costs and threats are more tightly associated than in this case study, where there is no clear spatial correlation between costs and threats (Sacre, Pressey, & Bode, 2019).

Our results illustrate some of the challenges that face conservation planners when attempting to design high-impact strategies. One key conclusion is that considering biodiversity data alone is unlikely to be sufficient, and that high-impact plans need to include data on threats. Although our analyses shows that relatively effective strategies can be designed using fairly simple and widely available datasets, such as satellite imagery of vegetation loss (Petrou, Manakos, & Stathaki, 2015; Xie, Sha, & Yu, 2008), strategies are likely to be greatly improved with better information on spatial patterns on a variety of threats. These could include more sophisticated spatial models of expected threats (e.g., Monteiro et al., 2020; Newburn, Berck, & Merenlender, 2006). For example, Newburn et al. (2006) developed a land-use change model that incorporated a variety of spatial factors, including slope, elevation, government zoning areas, and land-use types, among others. Such models are likely to be useful in designing threat prioritizations that are more effective than those used in the present analysis. Future analyses should also attempt to incorporate data on other types of threat, such as invasive species, pollution, and illegal harvesting, among others, on which data are less readily available (Joppa et al., 2016), but are likely to significantly influence the impact of planning strategies. However, regardless of the datasets and targets used by conservation scientists, of paramount importance is that prioritization strategies are tested within an impact framework. Although it cannot be expected of conservation practitioners to always develop sophisticated models of conservation impact, it can be expected that the strategies they choose to employ be supported by empirical evidence. Failure to do so, and reliance on familiar goals such as PA extent and representation, will inevitably lead to low-impact conservation.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

AUTHOR CONTRIBUTIONS

All authors participated in the design of the study. Edmond Sacre was responsible for data collection, analysis, and writing of preliminary manuscripts. All authors were involved in providing feedback on the analysis and revising the manuscript. All authors approved the final submission.

DATA AVAILABILITY

Data for this research are available at the Tropical Data Hub (https://tropicaldatahub.org/), under the catalog ID df2c6be9fd27f33a87f2e85d6e3a603e. Code to reproduce the analysis is available from the corresponding author upon request.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.