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Towards strategic offsetting of biodiversity loss using spatial prioritization concepts and tools: A case study on mining impacts in Australia

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ABSTRACT

Governments and industries increasingly use offsets to compensate for the unavoidable impacts of development on biodiversity. However, high uncertainty about the biodiversity outcomes of offsetting strategies has led to significant criticism in the academic and policy literature, while the ad-hoc application of offset rules within a region may lead to offsets favouring some species and communities at the expense of others. Here we explored opportunities to improve offsetting outcomes through strategic regional offset approaches, underpinned by concepts of complementarity and irreplaceability from the conservation planning literature, in comparison to more commonly used like-for-like approach. We assessed different offsetting strategies in the Hunter Valley, NSW, a rapidly developing region in Australia with an active mining industry. We quantified regional-level biodiversity losses arising from minimal to extensive mining expansion, along with species-specific impacts for 569 flora and fauna species, and prioritized areas for protection, restoration or both to offset the anticipated losses. Accounting for how well the offsets would complement existing protected areas, we compared the area needed for offsetting and the expected biodiversity outcomes among the different strategies. Our results highlight the benefits of a more systematic approach to offsetting in terms of an enhanced understanding of regional-scale impacts, more efficient identification of offset sites and improved biodiversity outcomes. Our approach encourages forward thinking about impending threats to, and opportunities for, biodiversity conservation and could serve as a template for strategic regional offset planning based on plausible scenarios of future biodiversity loss.

1. Introduction

Offsetting is being widely adopted in land-use planning in the attempt to settle the conflict between increasing human land-use needs and biodiversity conservation (Madsen et al., 2010). Offsets are intended to compensate for the residual, unavoidable biodiversity loss from development (ten Kate et al., 2004) with the philosophy that the compensations should match or exceed the anticipated harm ('no net loss', BBOP 2012). A plethora of offsetting mechanisms exist to date; however, the two most common mechanisms to directly compensate for the biodiversity value lost at an impact site are to protect existing habitats or to restore degraded sites elsewhere in the landscape (Bekessy et al., 2010; Maron et al., 2012). Both mechanisms aim to deliver direct, ecologically-equivalent gains in compensation for losses to achieve the no net loss status (Maron et al., 2012), as opposed to indirect compensation such as purely financial investment in biodiversity (Madsen et al., 2010). Offsetting by protecting and restoring existing habitats assumes that benefits will accrue through improving the condition of the targeted sites and increasing their security against other future losses ('averted loss').

With increasing popularity of offsetting schemes and programmes (Madsen et al., 2010), criticism of their functionality and usage has become more widespread. Concerns have been raised that offsetting programmes could act as an incentive for developers to shift their focus away from impact avoidance, leading to perverse outcomes where offsets are used to justify biodiversity losses without the ability to adequately compensate for these losses (Quétier and Lavorel, 2011; Moreno-Mateos et al., 2015-this issue). Shortfalls have been identified even in ecologically-equivalent offsetting programmes, which have been criticized for ill-defined objectives (Maron et al., 2012) and a lack of functional indicators to measure impacts and monitor outcomes (Quétier and Lavorel, 2011). Identifying the offsets required to compensate for loss typically involves 'like-for-like' indices of varying complexity that can combine multiple ecological variables (e.g. a hectare of a specific vegetation type needs to be offset by a hectare of the same vegetation type) (Madsen et al., 2010; Quétier and Lavorel, 2011; ten Kate et al., 2004). Such metrics act as currency in the transactions of trading-off one site for another, in the majority of cases aiming to identify offsets as similar as possible to the impact site. The like-for-like policy is strongly maintained because of the difficulty in valuing

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dissimilar habitats and ecosystems (ten Kate et al., 2004); however, for the same reason, such indices tend to be only a crude characterization of the ecological systems they represent. These indices are problematic for several reasons. The indexing metrics used to integrate multiple ecological components can be black-boxes that inhibit clear understanding of the impact on individual attributes and hence may lead to perverse ecological outcomes (McCarthy et al., 2004; Walker et al., 2009). The management objectives behind particular indicators are also often opaque, or not articulated at all (Maron et al., 2012), making the use of an index outside of its originally intended management context very risky. In most cases, the metrics are poor surrogates for biodiversity as a whole and for landscape-level ecological processes (Maron et al., 2012; Moreno-Mateos et al., 2015-this issue). Ad-hoc application of offset rules may therefore lead to poorly understood biodiversity outcomes at the regional scale. Outcomes may favour some species and communities at the expense of others, leading to failure in meeting regional biodiversity management objectives, such as maintaining the persistence of species or ecological communities.

The dangers of ad-hoc, rule or score-based site selection are well known within the field of reserve design, as they are known to result in inefficient reserve networks that do not capture the full range of the biodiversity features they aim to protect (Kirkpatrick, 1983; Pressey and Tully, 1994). The field of systematic conservation planning identifies the key principles of complementarity and irreplaceability (Margules and Pressey, 2000) to deal with this problem. The most cost-efficient way of building a reserve network that protects all biodiversity components (e.g. species or communities) is to select sites that *complement* each other in terms of the features they contain (Kirkpatrick, 1983). In practice, complementarity-based approaches are used to identify areas that will most efficiently add under-represented biodiversity features to the existing protected area network. Irreplaceability measures uniqueness of a site in terms of the biodiversity features it contains and is used to ensure that sites with rare biodiversity features, for which there are few or no alternative sites in the landscape, are prioritized in the site selection process (Pressey et al., 1994). Irreplaceability, when used in the conservation planning context, has no relevance to whether or not a particular ecological community contained in a site can be restored (replaced) in another part of the landscape (sensu Curran et al., 2014). By systematically identifying areas of high complementarity and irreplaceability, it is possible to improve the effectiveness and efficiency of conservation efforts (Margules and Pressey, 2000). This finding applies equally to offsetting as it does to reserve planning, where it has been most widely used to date. For example, using the irreplaceability concept in offsetting policy and practice could help to ensure that rare biodiversity features are not traded-away in favour of more common ones when identifying offsets, and to decide when a site cannot be offset. Concept of complementarity helps to recognize cases where offsetting impacts on common biodiversity feature by protecting or restoring habitat for more rare and threatened features provide greater biodiversity benefits (given that care is taken to avoid perverse outcomes such as the slow loss of originally common features, e.g. Regnery et al., 2013; Bull et al., 2015-this issue). A large number of freely available and widely used conservation planning tools implement complementarity and irreplaceability analyses in a conservation planning context, but thus far these have been rarely applied to offset analyses (Kiesecker et al., 2009; Moilanen et al., 2011; Overton et al., 2013).

Here we explore the benefits of applying principles of complementarity and irreplaceability in offsetting, by comparing options to offset the impacts of mining on 569 flora and fauna species across a region in south-east Australia. We outline a strategic, complementarity-based approach using common modelling and spatial prioritization software, in which the anticipated losses from development and gains from offsetting are quantified for each species. We then compare our approach to a more commonly used like-for-like approach, which is based on vegetation types rather than species distributions, and assess the biodiversity outcomes of different offsetting approaches under 20 mining scenarios. The primary purpose of this work is to demonstrate how conservation planning tools can be used to reveal the trade-offs in choosing any single offsetting approach, facilitating the assessment of both regional-scale and species-specific biodiversity impacts.

2. Material and methods

2.1. Study area

The Lower Hunter Valley, New South Wales, Australia covers approximately 430,000 ha with 60% covered in native vegetation (Fig. 1). The region contains features of national environmental importance, including a number of threatened species, both within and outside existing conservation areas (DECCW, 2009). The region supports a variety of land uses including open-cut coal mining, manufacturing industries, tourism and a large agriculture sector. Economically the Lower Hunter has a strong mining heritage, specifically for coal, and the current and pending coal mining titles cover approximately 21% (90,500 ha) of the region. Preliminary investigations indicate a significant overlap between new mining interests and areas of high biodiversity importance in the region (DECCW, 2009).

2.2. Species current and historic distributions

Occurrence data for species with more than 20 records within the Greater Hunter region were obtained from two online databases for 569 threatened species (36 amphibians, 289 birds, 61 mammals, 129 plants and 54 reptiles, Appendix A). Species distributions were modelled using MaxEnt (Phillips et al., 2006, version 3.3.3k) and a set of ecologically-relevant environmental variables describing aspects of climate, vegetation, topography and soils (Appendix A). MaxEnt models for each species were constructed using hinge features, with five-fold cross validation and taxa-specific sampling bias grids to account for potential spatial biases in the occurrence data (Kramer-Schadt et al., 2013). All modelling was undertaken at the scale of the Greater Hunter, using a 100 m grid cell resolution. Modelling at the broader scale enabled us to utilize more biodiversity data and avoid edge effects in the fitting data and predictions, increasing the robustness of SDM predictions. We used the average logistic output from MaxEnt to describe the current distribution of each species. In addition, to identify potential sites for restoration, we modelled the relative suitability of the currently cleared landscape for each species, assuming that restoration efforts at a given site would attempt to restore a vegetation community similar to historic vegetation patterns. We used data on estimated pre-European vegetation patterns, produced by NPWS (2000) using a decision tree model that combined current vegetation survey data with soil and topographical data (NPWS, 2000). We re-modelled species distributions, substituting variables of extant vegetation patterns with equivalent variables of pre-European vegetation patterns, to produce distribution maps that cover currently cleared but un-built-up areas in the region. All model outputs were clipped to the Lower Hunter and used in subsequent analyses. The assumption that currently cleared areas can be restored to provide habitat value for species is a controversial, but widely used assumption in many offsetting schemes throughout the world. It is beyond the scope of this paper to evaluate the voracity of that assumption. For a detailed treatment of how restoration uncertainty can be factored into conservation planning and offsetting analyses, see Moilanen et al. (2009).

2.3. Vegetation condition layer

A layer describing the condition of native vegetation (Fig. 1B) and anthropogenic disturbance was compiled using land use information (DECC, 2007) and the distribution of remnant native vegetation in the Lower Hunter (Cockerill et al., 2013). The original land use polygon

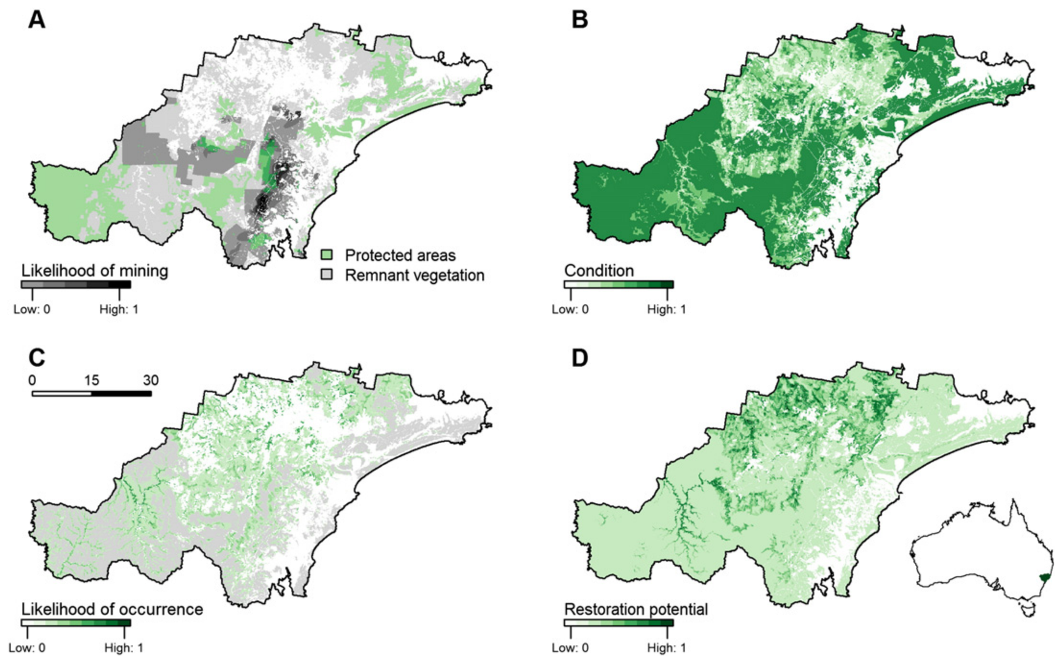

Fig. 1. The Lower Hunter Valley showing (A) coverage of remnant native vegetation, current protected areas and the estimated likelihood of mining activities. (B) The compiled vegetation condition layer covering all terrestrial unbuilt areas in the Lower Hunter. Panels (C) and (D) show the modelled extant distribution *N* of River Oak (*Casuarina cumminghamiana* subsp. *cumminghamiana*), and its estimated potential restoration return *N**, respectively. The location of the Hunter Valley is indicated on the inset map.

data (DECC, 2007) was first rasterized to 100 m resolution and reclassified into five condition categories (0, 0.25, 0.5, 0.75 and 1) indicating the intensity of disturbance (0 = cleared; 1 = intact) (Table A2). This condition layer was then further refined by overlaying it with a more detailed map of remnant native vegetation patches within the Lower Hunter (Cockerill et al., 2013).

2.4. Coal mining likelihood layer

There is little public spatial information about potential mining sites in the Lower Hunter. Therefore, we generated a surrogate layer using publicly available information obtained from MinView (NSW Resources & Energy, 2014) about exploratory drilling locations for coal, current titles, and applications for future titles that would allow for the development of coal mines. We used kriging to generate a density surface from exploratory borehole data, where a high density of boreholes was assumed to correlate with a higher probability of mining activity. We generated a second layer that ranked land parcels based on their current title status, with current mining titles considered more likely to be mined than mining title applications that had not yet been approved, followed by areas that had exploration-only titles. The density surface was then clipped to this title status layer and the density score in each pixel was scaled according to the corresponding mining title rank. This composite output was then scaled from 0 to 1, where areas of high borehole density within current mining titles represented a higher relative likelihood of future mining activity (Fig. 1A). We generated a range of potential mining scenarios that represented a gradient of mining pressure by selecting the top proportion of the landscape (5–100% in 5% increments) based on the relative likelihood of mining activity.

2.5. Spatial prioritization of offsets and assessment of their biodiversity outcomes

Using the principles of irreplaceability and complementarity, we analysed three hypothetical scenarios where mining impacts were offset by (1) protecting existing native vegetation, (2) restoring degraded or cleared land, or (3) a combination of protection and restoration (Fig. 2), taking into account how well species are currently protected by current protected area network (Fig. 1A). We measured the net biodiversity gains and losses of the offsets and explored the cost-effectiveness of the different scenarios, hereafter called 'strategic scenarios'. We chose the term 'strategic' to indicate that we are strategically considering the location of offsets on the basis of complementarity and irreplaceability, appreciating that there are many other ways to be strategic. To explore the relative efficiency of using complementarity-based offsetting approaches, we tested three additional scenarios: a (4) random scenario, which was otherwise identical to scenario 1 but the sites were selected randomly; and two like-for-like scenarios, mimicking current offsetting practices in NSW, where sites with same plant community types to the ones lost were either (5) protected or (6) restored. For each scenario we calculated the offsetting target and anticipated biodiversity outcomes over a gradient of mining pressure (area mined). For the strategic and random scenarios, the offsetting target was taken as the area required to either protect or restore an equal amount of biodiversity value to that lost due to mining, measured as the summed relative habitat value (based on SDMs) of sites overlapping with potential mining across all species. For the two like-for-like scenarios the target was taken as the area needed to protect or restore exactly the same amount of plant community types lost to mining. The total biodiversity outcomes of all offsetting schemes were compared by calculating the average proportion of species original distributions remaining in the landscape after offsets are allocated. We also measured how much the offsets in each scheme would improve the protection of species, assuming that in all scenarios the identified offset sites would be added to the existing protected area network.

We used the spatial prioritization software, Zonation v.4.0 (Moilanen et al., 2012) to prioritize offsets based on principles of complementarity and irreplaceability. Zonation is a maximum-coverage type spatial prioritization tool used to select areas that maximize representation of multiple biodiversity features (from now on called species) within budgetary or area constraints (see Appendix B for details). Although in this study we focus on species, we note that these features could

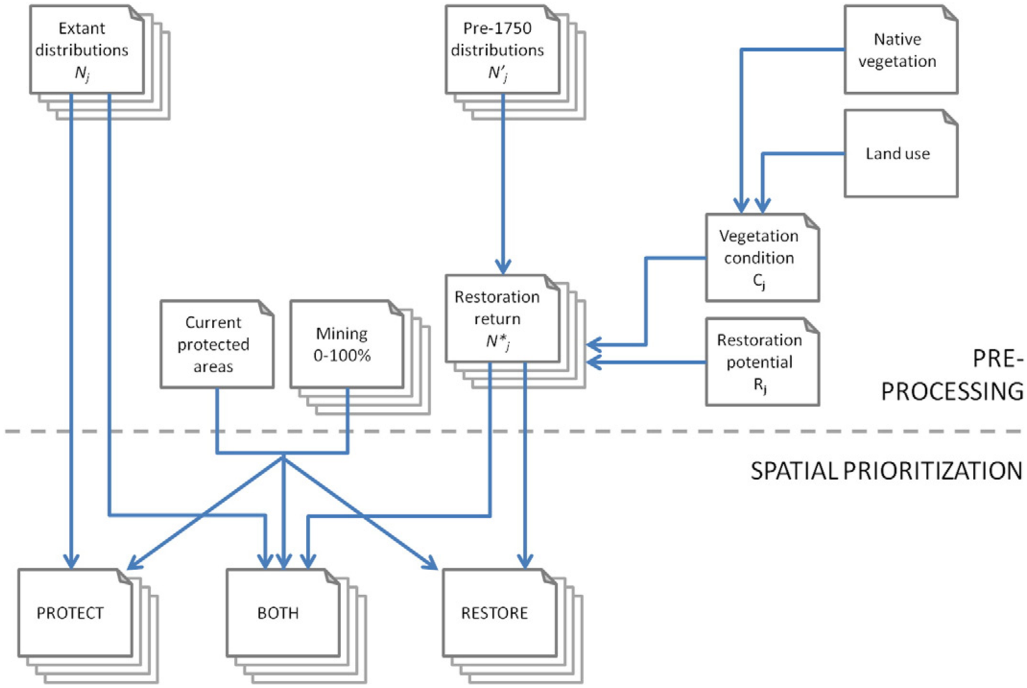

Fig. 2. Schematic diagram showing the data inputs of the spatial prioritization of offsets.

represent any type of biodiversity component, such as communities, vegetation types, genes or phylogenetic diversity (e.g., Pollock et al., 2015).

Priorities for the strategic offsets were selected using the inbuilt features of Zonation, which allow consideration of habitat condition and potential changes in condition depending on whether a conservation action is implemented or not (Moilanen et al., 2011). For each species we used both the extant (N_{ij}) and pre-European (N'_{ij}) distribution with different weights to define which sites should be protected and which restored. In the first scenario, where offsets only protect existing habitat, species extant distributions (N_{ij}) were used to identify offsets, weighting all species equally ($w_j = 1$ for all species).

For the second scenario, where strategic offsets only restored currently degraded or cleared areas, we created a potential restoration returns layer, N^*, for each species and used these to guide the prioritization (Fig. 1D). These layers give the estimated difference in outcome that restoration delivers for each species at each site compared to no restoration. The layers were created from species pre-European distributions (N'_{ij}), current local vegetation condition (C_i, Fig. 1B) before restoration, and estimated restoration potential (R_i), which gives the relative increase in local quality after restoration actions have been carried out. Ideally, information on restoration potential should be based on empirical studies of ecosystem and species responses to management, however, such information is scarce and uncertain (Curran et al., 2014). For the purpose of this study, we used a simplifying assumption that restoration at each site would be successful in returning the condition of the site to its original value. We acknowledge that such an outcome is highly uncertain but allows us to illustrate the difference between offsetting strategies. Hence, we used one plus the complement of condition as the estimate of restoration potential (value 1 indicating no potential and value >1 indicating increasing potential), and defined the final restoration return N^*_{ij} as:

$$N^*_{ij} = |1 - R_i| C_i N'_{ij}.$$

In scenario two, we used these species-specific restoration return layers to prioritize areas for restoration. Zonation weights the species restoration return layers (N^*_j) based on their retention value (Moilanen et al., 2011) (i.e., how much difference restoration is anticipated to deliver compared to no restoration). The logic is that scarce management resources should be targeted at species whose status is improved by action, either by improving their currently undesirable situation or by averting a potential threat. This weighting, in addition to the species weights w_j, is:

$$w^r_j = \frac{\sum_i |1 - R_{ij}| C_{ij} N'_{ij}}{\left(\sum_i |1 - R_{ij}| C_{ij} N'_{ij}\right) + \left(\sum_i C_{ij} N_{ij}\right)} w_j$$

where the numerator is the summed restoration returns and the denominator is the expected final condition for a species after restoration actions have been taken.

In the third scenario, where offsets were selected by either protecting existing habitat or restoring new habitat, we used both species extant distributions (N_{ij}) and their potential restoration returns (N^*_{ij}) to prioritize sites. The balance between protection and restoration was controlled with a scaling parameter β, which scaled the weights w^r_j, of all restoration return layers relative to the extant distribution layers. When β is zero, only existing habitats are prioritized for protection and increasing β values increase the emphasis given to restoration over protection. Setting β is subjective and case-specific but it can be used to reflect uncertainties associated with the estimated returns from restoration actions. Setting lower values is recommended when the benefits from restoring degraded habitats are more uncertain. Ideally this parameter would be based on empirical estimates of restoration success probabilities. After initial testing of a range of β values we used $\beta = 5$ for all analyses in scenario 3 as this parameter value meant that the relative median weight of species potential restoration returns (N^*_{ij}) was about half of that of their extant distributions (Fig. A1). The area needed to offset mining impacts in this scenario was not sensitive to the choice of β (Fig. A1).

We mimicked the current offsetting practices in NSW (OEH, 2014), and used a like-for-like strategy to identify offsets for the plant communities lost under mining, either within extant vegetation (protect) or currently degraded or cleared lands (restore). We used the pre-European vegetation patterns (NPWS, 2000) to map the distribution of 58 plant community types in the Lower Hunter, converting each plant community type to 100 m resolution raster layer. We then quantified the amount of each plant community type lost to mining and prioritized the remaining unprotected area to find offsets that returned exactly the same amount of each impacted plant community type. For the like-for-like approaches we used the Marxan software (version 2.4) (Ball et al., 2009) to prioritize areas for offsets. Marxan is a target-based planning tool which is ideal for identifying area/cost efficient like-for-like options (see Appendix B for details about Marxan and the settings used to find offset locations).

After prioritization, in each scenario, we identified the top priority areas that satisfied the offsetting requirements. For strategic and random scenarios these were sites that returned the summed proportion of species original distributions lost to mining. For the like-for-like scenarios these were sites that met the targets for all vegetation types. If some targets could not be met (i.e., there was not enough area to protect or restore a particular vegetation type to match the lost area), the top priority areas were defined as sites that met all achievable targets plus any remaining areas for those vegetation types for which targets could not be achieved. In all analyses we initially assumed that protecting or restoring any given site is equally costly, the final costs in this case being dictated by the number of cells needed to meet the offsetting requirements. To explore the impact of costs we reran the strategic scenarios assuming that the cost of restoring a cleared site (pixel) was three or ten times more than protecting an intact site of same size (See Appendix B for details on how costs are incorporated to the prioritization) and measured the increase in cost to offset the mining impacts. Restoration costs of degraded sites in both cost scenarios were dependent on the starting condition of the pixel and scaled between the cost of cleared and intact sites.

3. Results

Overall, the model performance of the SDMs was high, with a mean cross-validated AUC value of 0.88 ($\pm$0.003) for all taxonomic groups (Table A3). Models with AUC values of 0.7 or greater were considered informative (Swets, 1988). Mining all existing and pending coal titles in the Lower Hunter would clear 22% of native vegetation cover, including 8% within currently protected areas. The average impact on species distributions increased linearly as the proportion of titles mined increased from 5% to 100% (Fig. 3A). When 5% of mining titles were developed, the average loss in species distributions was estimated at 1.4%, increasing to 21.9% when all mining titles were cleared. Variation among species was high, with some species notably more impacted, such as the superb lyrebird (*Menura novaehollandiae*) which would lose almost 60% of its local distribution if all mining titles were cleared. Within the vegetation communities, the average loss ranged from 0.6% to 14.9% with increasing levels of mining, the most impacted community (redgum rough barked apple forest) losing up to 88% of its current extent.

Using the strategic approaches the area needed to offset mining impacts varied between 4360 and 64,050 ha (1.2–17.6% of the Lower Hunter) depending on the mining pressure. In scenario 3, where offsets could be either protected or restored, restoration was emphasised at low levels of mining and shifted towards protection at high levels of mining (Fig. 4). In all three scenarios, the area offset was larger than the area mined up to ~35% of the titles, above which offsets for the anticipated losses could be found in a smaller area than the area mined. The difference in offset size between the three strategic offsetting approaches was generally small. Unsurprisingly, when protection

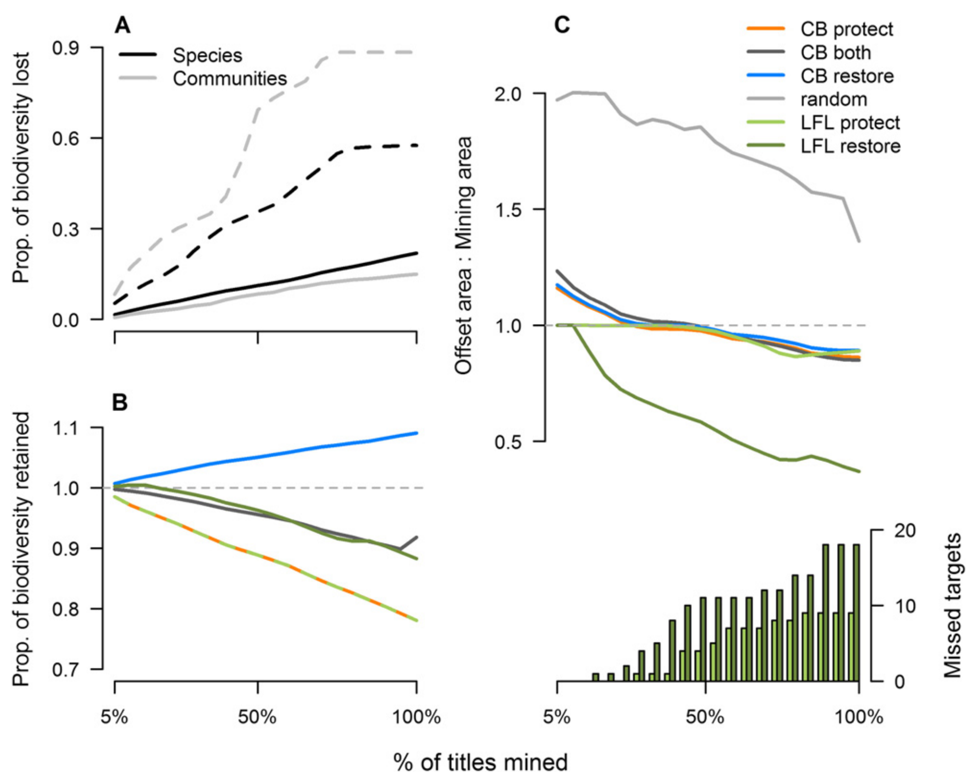

Fig. 3. (A) The average (solid) and maximum (dashed) anticipated loss in the distributions of species and vegetation communities over a gradient of titles mined. (B) Final biodiversity values retained after offsetting. This gives the average proportion of species original (pre-mining) distribution that is left in the landscape after mining impacts have been offset. The grey dashed line indicates the starting point, the level of original biodiversity values before any mining. Note that lines for SO protect, LFL protect and random are overlaid. (C) Ratio of offset area to area mined. Values above the grey dashed line indicate that more area is needed to offset than what is being mined. Values below the line indicate that a similar amount of the biodiversity value lost to mining can be offset elsewhere in the landscape with a relatively smaller area. The bars under the lines give the number of plant community types for which offsetting targets were missed in the two like-for-like scenarios. Like-for-like offsetting through restoration (LFL restore) struggled to find enough suitable vegetation types to meet the offsetting targets and therefore, used notably less area for offsetting. However, even at low levels of mining where targets are met (C) the average benefits for biodiversity (B) were lower than when offset sites for restoration were selected strategically (SO restore).

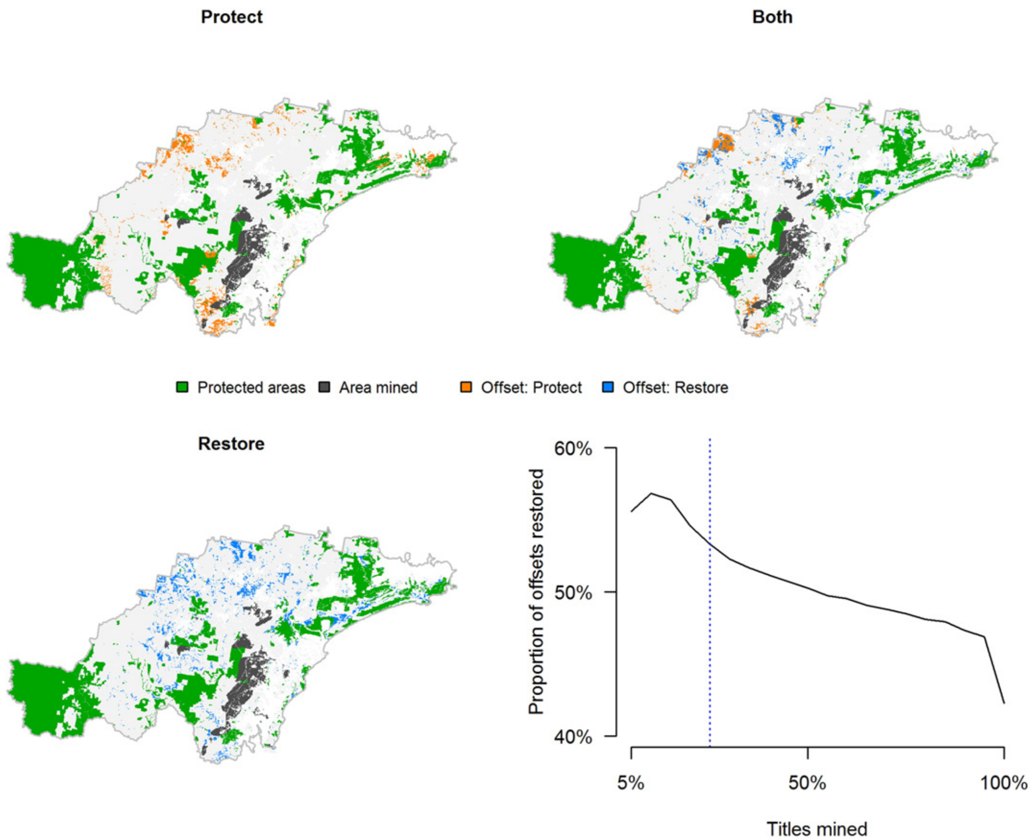

Fig. 4. Offsetting under the three strategic scenarios (protect, restore, both) when the most probable 25% of titles are mined (dark grey). Protection and restoration offsets are shown in orange and blue, respectively and existing protected areas are in green. The lower right panel displays the proportion of offsets in scenario 3 (both) that are restored over a gradient of titles mined, with the dotted vertical blue line denoting the proportion of titles mined in the mapped scenarios.

was the sole offset type, there was a 22% cumulative loss of original biodiversity value, whereas offsetting entirely with restoration was the only strategy that could potentially maintain or improve biodiversity value compared to pre-development conditions (Fig. 3C). Although the offsetting scenarios that included restoration required more area, their benefits outweighed the lower area requirement of only protecting sites when the action of restoring or protecting were assumed to have equal cost. When restoring a cleared site was three or ten times more costly than protecting extant vegetation, offsetting through restoration required, on average, 2 or 2.6 times more money to achieve the offset targets than offsetting by protection at any level of mining, respectively. The combination of protection and restoration would need 1.7 and 2 times more money, on average, under the three and ten times higher restoration costs, respectively.

Protecting habitats using principles of complementarity and irreplaceability was clearly more area-efficient in offsetting mining impacts than selecting offsets for protection randomly, which required 1.6–1.9 times more area at any level of mining (Fig. 3B). Protecting habitats using the like-for-like approach required less area to offset the impacts than the complementary approaches at very low levels of mining, whereas restoring like-for-like habitats required always clearly less area. However, the apparent area-effectiveness of like-for-like approaches was driven by the increase in missed targets, as an increasing number of plant community types became un-offsettable when 10% or more of the mining titles were cleared (Fig. 3B). For up to 9 and 18 plant communities (out of 37 impacted by mining) there was not enough potential habitat to protect or restore, respectively, in order to offset the increasing mining impacts using the like-for-like approach, effectively reducing the size of selected offsets. Consequently, even when restoring offset sites using the like-for-like approach it was not possible to meet the no-net-loss target at intermediate and high levels of mining but there was a mean loss of up to 8.5% in the vegetation community extent despite restoration efforts. Furthermore, the failure to meet the restoration offset targets translated into a cumulative 11.3% loss in biodiversity values measured across species (Fig. 3B).

Assuming that the offsets would be added to the existing reserve network, the amount of protected habitat could on average be 1.05–1.8 times higher than current levels depending on the level of mining and strategy chosen. The strategic offsetting strategies tended to improve species protection more efficiently at all levels of mining, the difference increasing with increasing mining (average increase in the amount of habitat protected after offsetting in each mining scenario: SO Protect: 10–72%; SO Both: 10–81%; SO Restore: 7–75%, LFL Protect: 5–60%; LFL Restore: 5–25%, Fig. 5). Due to the high number of missed targets and smaller offset size, restoring like-for-like habitats returned least benefits to species protection, up to 56 species having less habitat protected after offsetting at high levels of mining (Fig. 5). Protection of like-for-like habitats also missed targets at high levels of mining, but resulted in roughly same offset area as the strategic protecting strategy (Fig. 3C). Protection of complementing sites, however, improved species protection by 10%-points more than protection of like-for-like sites, the worst-off species gaining 13.7% more protection under the strategic approach, and losing 4% under the like-for-like approach (Fig. 5).

Further analyses revealed marked differences in the way that individual species benefited from offsetting under the different strategies. Even within the strategic scenarios, where offsets were selected to return equal biodiversity values across all species, offsetting through protection tended to return intermediate gains for a wider range of species, while offsetting through restoration returned high gains for a smaller number of species (Figs. 5, and A2). With increasing levels of mining, marginally more species would get greater benefits if offsetting

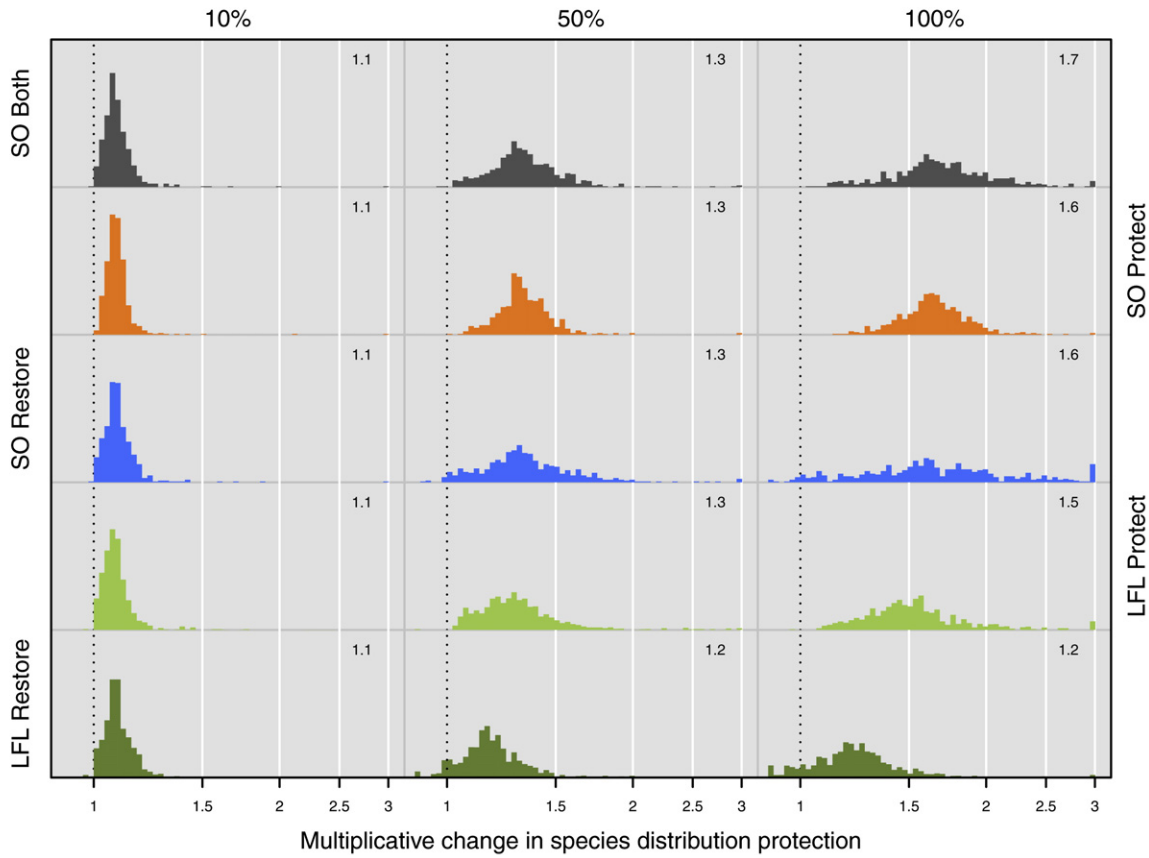

Fig. 5. Change in proportion of habitat protected after offsetting at 10%, 50% and 100% level of mining in comparison to original protection (before mining). Bar plots show the relative frequency of change in protection across all species when offsets are implemented through strategic protection (orange), restoration (blue), or both (grey), or when impacts are offsetted using like-for-like approach to protect (light green) or restore (dark green) habitats. Values right from the grey vertical line indicate that species have more of their habitat protected after offsetting in comparison to pre-mining situation. Numbers on the top-right corner of each plot gives the median value across the entire species pool. Species with more than three times or less than nine-tenths change in protection are allocated to the largest and smallest bins, respectively.

was done through protecting existing habitats than if they were only restored. For the majority of species both scenarios 1 and 2 are likely to compensate mining impacts in a relatively similar way. However, for some species, such as the river oak (*Casuarina cumminghamiana* subsp. *cumminghamiana*) and the nationally listed, critically endangered North Rothbury Persoonia (*Persoonia pauciflora*), selecting one scenario over another would result in clear differences in expected offsetting outcomes (Fig. A2). How well offsetting would compensate species for mining impacts and improve their level of protection is dependent not only on the selected offsetting strategy, but also on the level of mining pressure, species current level of protection and the proportion of their historical distribution that remains. For species like the North Rothbury Persoonia, restoring larger offset areas as compensation to increasing mining pressure brings few gains as there are no more suitable areas to restore for the species (Fig. A2). Increasing mining pressures themselves also diminish opportunities to offset impacts. For example, outcomes for the broken back ironbark (*Eucalyptus fracta*) in scenario 1 (protect) are decreased by a half as the level of mining increases beyond 85% and the expanding mining areas clear almost half of the species' remaining unprotected distribution (Fig. A2).

4. Discussion

Ad-hoc application of poorly designed and monitored offsetting schemes that are used in site-by-site assessments is unlikely to halt biodiversity loss (Quétier and Lavorel, 2011). Here we have shown how spatial prioritization tools can be used to compare the benefits and costs of different offsetting options across multiple species in a large landscape, and to create cost-efficient offsetting strategies that minimize losses for all components of biodiversity that can be mapped. Our approach also allows managers to track individual winners and losers under any given impact scenario, and thereby avoid inequitable or otherwise unacceptable outcomes (Bull et al., 2015-this issue). Comparing different types of offsetting approaches that have different objectives (minimizing landscape rarity for all species versus maximizing like-for-like protection), and use different currencies is challenging. Our findings are not inherently surprising, in that they show complementarity- and irreplaceability-based offset site selection leads to better conservation outcomes because they target the rarest and least protected species. On the other hand, a like-for-like strategy that trades on hectares of vegetation types tends to do a better job of replacing what is lost in any individual development. However, what we have achieved here is quantifying the benefits to biodiversity by being strategic with offsetting, rather than using the more traditional like-for-like approach (Figs. 3A and 5).

Our analysis shows that the impacts of mining, and options to offset those impacts, vary greatly between species in the Lower Hunter (Figs. 3A, 5 and A2), emphasising the importance of explicitly assessing impacts on species and threatened communities and not relying on generic indicators such as vegetation type and condition. Species-level differences in apparent impacts of mining are dictated by species' current and past distributions, the extent of the impact, and how much offsetting potential remains in the landscape after development. In our study, analysing coarse surrogates such as the loss of vegetation area and type would not have revealed the extent of impacts on particular species. Undertaking like-for-like offsetting of vegetation types would result in overall net losses of up to 10% of biodiversity value (measured as species distributions) under the assumption of successful restoration, while strategic offsetting led to a 10% increase in biodiversity value (Fig. 3B). Both will do much worse if restoration is less than 100% successful, however uncertainty about the success of restoration undermines both strategies equally in this analysis.

In Lower Hunter, the most likely mining sites based on current exploration activities require 16–23% more area to compensate for lost species distributions (Fig. 3C), indicating that these mining areas have high habitat value for species that are relatively difficult to replace elsewhere in the landscape. For vegetation communities, replacements for lost habitat at low levels of mining could be found for smaller area, implying that using vegetation communities as offsetting currency does not necessarily correctly portray the level of impact and offsetting need for the species they host. We underline that in all scenarios explored in this work the area needed to truly offset mining impacts is likely to be greater than estimated here because our analysis largely ignores restoration uncertainty (Curran et al., 2014; Moilanen et al., 2009).

The like-for-like approach struggled to meet targets under moderate to high levels of mining. This raises the question of how unmet targets should be properly, and transparently, compensated. One approach is to extend offsets to outside the region; however, it is not always reasonable to assume that suitable offsets can be found in nearby regions. In this study the like-for-like approach was based on a relatively coarse plant community classification currently used as the basis of the official offsetting scheme in NSW (OEH, 2014). Using a more, or less, detailed community classification or other environmental criteria as currency for offsetting could make the approach more successful in meeting the targets.

Using quantitative and transparent planning tools allows comprehensive assessment of trade-offs between different offsetting strategies. Offsetting through the protection of existing habitats tends to be cheaper and more certain than restoration-based offsets. However, to satisfy the additionality principle of offsetting (ten Kate et al., 2004), it is assumed that the area being protected as an offset would otherwise be (eventually) lost to land clearing or degradation. Nonetheless, offsetting by protection (or avoided loss) will always fail the 'no-net-loss' objective. In the case of the Lower Hunter, offsets that avoid future losses by protecting existing habitats lead to the average loss of up to 22% in species distributions, irrespective of the offsetting approach. Offsets based on restoration, on the other hand, are the only way to achieve true no-net-loss or net gain in biodiversity value, and in the Lower Hunter have the potential to maintain or even improve biodiversity values at a regional scale. However, there are serious concerns about the capacity of restoration to bring biodiversity benefits at broad spatial scales and within reasonable time-frames (Curran et al., 2014; Vesk et al., 2008; Moreno-Mateos et al., 2015-this issue). Our work does not attempt to address this contentious issue. The viability of restoration as an offsetting approach has no bearing on our findings about the relative merits of complementarity-based and like-for-like offsetting strategies. For any individual offsetting activity, it is possible to accommodate extra complexities such as restoration time-lags (Bull et al., 2015-this issue), costs and uncertainty using existing software (e.g., Moilanen et al., 2009), though this can add significant complexity to analyses. For the case of offsetting mining impacts in the Lower Hunter, a balanced portfolio of both protection and restoration might provide a useful middle ground, where risks and opportunities are balanced in a manner that is suitable to managers and stakeholders and which allows better tailoring of offsets according to species needs (Fig. A2).

In this work we have used static species distributions and maps of threatened ecological communities to represent regional biodiversity. This approach ignores ecological processes such as population dynamics and species interactions. The spatial prioritization tools used in this case study offer some options to include population dynamic processes in the analysis such as species-specific movement and connectivity requirements (Lehtomäki et al., 2009; Moilanen and Wintle, 2006). Under such an approach, restoration activities could be targeted towards areas close to existing species occurrences, to decrease the level of fragmentation, increase landscape connectivity and improve the likelihood of successful habitat recovery.

By considering both complementarity and irreplaceability when identifying potential offset sites, our approach ensures that species and ecological community across the entire landscape are explicitly considered. The modelling approach presented here uses readily available tools to identify the type, size and location of offsets required to compensate for biodiversity losses due to development under three different offsetting mechanisms (protection, restoration or a combination of the two). It addresses the non-trivial challenge of how to reconcile and integrate protection and restoration options when identifying offsets that provide the best regional biodiversity outcomes. While our approach may add an additional level of analytical complexity to offsetting policy and practice, there is substantial benefit in using a transparent and repeatable process to help decision makers quantify biodiversity losses due to proposed development and tailor offsets to the specific objectives of managers and stakeholders. Our approach encourages forward thinking about impending threats to, and opportunities for biodiversity conservation. It could serve as a template for strategic regional offset planning based on plausible scenarios of future biodiversity loss.

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