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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