

Biodiversity Offsets Can Be a Valuable Tool in Achieving Sustainable Development

Developing a Holistic Model for Biodiversity Offsets That Incorporates Environmental, Social and Economic Aspects of Sustainable Development

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Abstract

The interpretation and use of biodiversity offsets in planning and development is a contentious issue because they rarely encompass each of the environmental, social and economic aspects of sustainable development. While currently agreed best practice for biodiversity offsets includes consideration of scope, scale, location, timing and duration, and monitoring, current literature on these components does not consider all aspects of sustainable development. Furthermore, much of the current agreed best practice focusses on the design of biodiversity offsets, without consideration of ongoing management or end-of-life. This manuscript reviews current best practice for biodiversity offsets, giving consideration to the environmental, social and economic aspects of sustainable development. In particular, we report that consideration of cost and risk is key and the use of planning frameworks, bonds and advanced offsets could mitigate these risks and allow for long-term success. Following this approach, a holistic model for design, implementation and ongoing management of direct biodiversity offsets that balances all aspects of sustainable development is presented.

Keywords: biodiversity, environmental management, environmental planning, natural resource management, offsets

1. Introduction

Biodiversity offsets are defined by the Business and Biodiversity Offsets Programme (BBOP) as “measurable conservation outcomes of actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken” (BBOP, 2019). To implement, the BBOP (2012) categorises the elements that biodiversity offsets should contain, which include scope (including type of compensatory activities), landscape interaction (scale), location, and implementation (including monitoring, management and reporting). However, as biodiversity offsets are often used to balance the loss from development with conservation gains (Fallding, 2014; Maron et al., 2016), they should also be aligned with the principles of sustainable development, and include not only the environmental, but the social and economic aspects of the ecological community as well (Abdo, Griffin & Kemp, 2019). Here sustainable development is defined as “development that meets the needs of the present without compromising the ability of future generations to meet their own needs” (International Institute for Sustainable Development, 2017). There are three key aspects to sustainable development that must be considered in balance to ensure that natural values (biodiversity, ecosystem services and ecosystem function) are not compromised: environmental, social and economic (Gibson, 2009; Moldan & Dahl, 2007; International Institute for Sustainable Development, 2017; Macintosh, 2015).

Biodiversity offsets are intended to be implemented with consideration to all three aspects of sustainable

development (Abdo, Griffin, Kemp, & Coupland, *in prep*; BBOP, 2012; MacIntosh, 2015). While much has been written on the design of biodiversity offsets (Bull, Suttle, Gordon, Singh & Milner-Gulland, 2013; Carreras Gamarra, Lassoie, & Milder, 2018; Gardner et al., 2013; Quétier & Lavorel, 2011), this previous work has predominantly focussed on the environment, excluding the social (Bidaud, Schreckenberger, & Jones, 2018; Gibbons, Macintosh, Constable and Hayashi, 2018; Githiru et al., 2015; Jacob, Buffard, Pioch & Thorin, 2017; Macintosh, 2015; Nijnik & Miller, 2017; Scholte, van Zanten, Verburg & van Teeffelen, 2016; Takacs, 2018) and/or economic (Benabou, 2014; Fallding, 2014; Jacob et al., 2017) aspects of sustainability, leading to inequalities and an inconsistent approach (Abdo et al., 2019; Jacob et al., 2017). Therefore, to ensure biodiversity offset requirements compensate for all aspects of sustainable development, a holistic model, incorporating natural values for design, implementation and 'end-of-life' phases, is needed. In particular, biodiversity offsets should address the Sustainable Development Goals that provide "strategies that build economic growth and address a range of social needs including education, health, social protection, and job opportunities, while tackling climate change and environmental protection" (United Nations, 2019).

This manuscript will seek to address this imbalance by: i) firstly providing a review of recommended best practice for the key elements of biodiversity offsets - scope, scale, location, timing and duration, and monitoring and measurement; ii) secondly, applying these aspects to the considerations of sustainable development; before iii) finally developing a holistic model for biodiversity offsets that balances all aspects of sustainable development. Note that the scope of this manuscript is restricted to aspects of biodiversity offsets that are chosen as part of offset design, implementation and ongoing management. Other considerations, such as counterfactual scenarios and environmental economics/metrics, that are chosen prior to this and generally as part of the assessment of impact from a development have not been explored.

2. Scope of Biodiversity Offsets

The scope of biodiversity offsets defines the aspects that will be offset and dictates the conservation gains that are to be achieved within a defined timeframe, thus identifying the expected ecological equivalence/no net loss (Bull & Brownlie, 2015; Carreras Gamarra et al., 2018; Gardner et al., 2013; Quétier & Lavorel, 2011). It is important that inclusions within the scope are broad enough to not only ensure that all key attributes required to secure adequate compensation are captured, but that the conservation gains and timeframe are also achievable. However, the scope of biodiversity offsets is often too narrow to effectively capture all environmental, social and economic concerns (Abdo et al., 2019; Fallding, 2014; Ghosh, 2015; Gibbons et al., 2018; Reyers et al. 2013; Takacs, 2018). For example, Gibbons et al. (2018) 10-year review of biodiversity offsets in New South Wales, Australia, found offset programs insufficiently considered the social aspects of intergenerational equity and the inherent value of different habitat types. Ghosh (2017) reported that the compensatory afforestation program in India was narrow in scope, focussing only on numerical valuations of forest type and region, ignoring inherent biological, spatial and social values of the ecosystems impacted. Similarly, Birkeland and Knight-Lenihan (2016) found that the scope of a biodiversity offset in New Zealand focused on compensation for species removed or directly impacted at the development site, without consideration of the impact from other stages (such as transportation, storage or construction) of the development life-cycle; which could result in further environmental and social impacts that remained unaccounted for.

The scope of biodiversity offsets should include assessment of not just species and habitats, but also ecological processes (Bigard, Pioch & Thompson, 2017; Gardner et al., 2013; Pilgrim et al., 2013), ecological function (Bigard et al., 2017; Bull et al., 2013; Gardner et al., 2013; Gonçalves, Marques, Soares & Pereira, 2015; Kiesecker et al., 2009; Moreno-Mateos, Maris, Béchet & Curran, 2015; Pilgrim et al.; 2013) and genetic variation (Bigard et al., 2017; Bull et al., 2013; Gonçalves et al. 2015; Moreno-Mateos et al., 2015), to ensure it is broad enough to provide adequate compensation. In addition, the scope should also include the potential impacts of climate change, both in terms of site selection and the potential effect of climate change on conservation measures (McDonald, McCormack & Foerster, 2016). However, the choice of ecological functions used for biodiversity offsets can be controversial from a social perspective, as some services (e.g. wetlands) are beneficial for some members of the community, while other aspects (e.g. wetlands harbouring mosquitoes) can have a negative impact on other community members (Moreno-Mateos et al., 2015). The type of compensatory measure, method of determining ecological equivalence, and the choice of biological indicators must be adequately considered for all aspects of sustainable development, in order to resolve these conflicts and ensure that the scope of biodiversity offsets is effective to compensate for the natural values (biodiversity and ecological processes, functions and services) impacted.

2.1 Type of Compensatory Measure

There are several types of activities that are considered appropriate as compensatory measures for biodiversity offsets. These are typically categorised as either indirect or direct offsets. Indirect offsets (also known as ‘other compensatory measures’) are “actions that do not directly offset the impacts on the protected matter, but are anticipated to lead to benefits for the impacted protected matter” (Department of Sustainability, Environment, Water, Population and Communities, 2012) and include knowledge acquisition and scientific research programs, as well as compensatory packages (Fallding, 2014; Jacob et al., 2017). In order to be considered appropriate, indirect offsets need to provide measurable biodiversity gains (Gardner et al., 2013). As the link between indirect offset activities and measurable biodiversity outcomes is not always clear, regulators usually require direct offsets over indirect offsets (Niner, Milligan, Jones & Styan, 2017). Therefore, this review will focus on direct biodiversity offsets.

Direct offsets are defined as “those actions that provide a measurable conservation gain for an impacted protected matter” (Department of Sustainability, Environment, Water, Population and Communities, 2012), and include habitat restoration (Gardner et al., 2013; Maron et al., 2012; McDonald et al., 2016) and management interventions to prevent loss (termed averted loss) (Fallding, 2014; Gardner et al., 2013; Jacob et al., 2017; Maron & Louis, 2018; Moilanen & Kotiaho, 2018). Biodiversity offset markets (including payments for ecosystem services) are also often considered direct offsets, as, although compensatory conservation activities are undertaken by third parties on behalf of a developer, they usually result in habitat restoration.

Habitat restoration biodiversity offsets rely on conservation activities that improve habitat quality and/or extent as a compensatory measure (Maron, 2012; McDonald et al., 2016). Habitat restoration biodiversity offsets should only be implemented where natural values can be explicitly defined, there is sound scientific evidence that restoration will be successful, and time lags and uncertainties are effectively accounted for (Maron et al., 2012). Habitat restoration offsets can ensure no net loss (Maron & Louis, 2018) but have been shown to have unpredictable costs and a lower likelihood of success. For example, Bekessy et al. (2010) reported that restoration projects are usually associated with time lags and uncertain outcomes, which often leads to loss of biodiversity. Maron et al. (2012) reported low success for restoration projects and, when revegetation occurred in a highly degraded area, the resulting restored ecosystem rarely reflected what was intended. Similarly, Bullock, Aronson, Newton, Pywell & Rey-Benayas (2011) meta-analysis of 89 restoration projects across a range of different ecosystems found that restored areas only provided on average 86% of the biodiversity and 80% of the services associated with reference ecosystems.

Averted loss biodiversity offsets are those that involve the maintenance and/or protection of sites that would otherwise be under threat (Maron, 2012; McDonald et al., 2016; Moilanen & Laitila, 2015). Averted loss biodiversity offsets are only able to halt decline and cannot offer no net loss or net gain, despite being lower in cost and easier to implement than habitat restoration offsets (Gibbons et al., 2018; Maron, 2015; zu Ermagassen, 2019). While habitat restoration biodiversity offsets are often preferred over averted loss biodiversity offsets (Githiru et al., 2015; Moilanen & Kotiaho, 2018), averted loss biodiversity offsets have the advantage in that they are lower cost, easier to implement (Maron, 2015), and can mitigate uncertainty (Maron et al., 2012). However, as averted loss biodiversity offsets aim to halt decline, rather than providing explicit biodiversity benefits, they are only appropriate where there is a substantial and certain ongoing or imminent threat to the biodiversity (Bidaud et al., 2018; Gardner et al., 2013; zu Ermagassen, 2019), the predicted loss of biodiversity is low and it is not critical that biodiversity offsets achieve their intended outcomes within a short period of time (Gibbons et al., 2018). Both habitat restoration and averted loss biodiversity offsets should ensure that compensatory activities provide equivalent ecological benefits for the natural values impacted to ensure that the principles of sustainable development are not compromised.

Biodiversity offset markets are a cost-effective solution (Benabou, 2014; Simpson, de Vries, Armsworth & Hanley, 2017) that can ensure no net loss/net gain of natural values. They provide a market for conservation activities undertaken by third parties, such as landholders (Bull et al., 2013; van Teeffelen et al., 2014). These activities generate credits that are then purchased by developers as compensation. Credits created by biodiversity offset markets have a value that is determined by future supply (Ozdemiroglu, Kriström, Cole, Riera & Borrego, 2009) and the design of the offset itself (Coggan, Buitelaar, Bennett & Whitten, 2013). This supply is in turn influenced by regulatory requirements, meaning that governments have a large, although often indirect, role in the development and maintenance of biodiversity offset markets (Coggan et al., 2013). In order to develop functional markets, biodiversity offsets need to have clear, transparent and specific requirements, stated compensatory activities and clear definitions around duration of impact, in addition to adequate numbers of buyers and sellers (Godden & Vernon, 2003).

Biodiversity offset markets provide an incentive for conservation, and thus may be influential in changing the behaviour of landholders (Filoche, 2017) and enabling governments to achieve conservation goals at a lower cost (Kleining, 2017). Biodiversity offset markets can also allow biodiversity offset gains to be achieved in advance of development impacts (Bull et al., 2015; Ozdemiroglu et al., 2009; van Teeffelen et al., 2014), can consolidate small offset projects into a larger project with value greater than the sum of smaller offsets (Benabou, 2014), while simultaneously enabling savings and efficiencies for regulators (Kormos, 2015). Biodiversity offset markets are valuable for developers as offset related costs are predictable and therefore the responsibility for managing an offset site can be delegated. Biodiversity offset markets can also provide an opportunity for communities to become more vested in the decisions around developments and their associated offsets, as well as potentially providing opportunities for the community to become offset providers and for the financial incentives of offsets to flow back into the community.

There is a risk, however, that biodiversity offset markets may result in further simplification of ecosystem measures, ultimately resulting in inadequate compensation of biodiversity, ecosystem function and/or ecosystem services. While complex banking schemes may ensure better ecological equivalency, the higher transaction costs of such schemes are likely to lower the potential financial gains from the trade (Simpson et al., 2017), which could impact on provider participation and ultimately affect the usefulness of the banking scheme overall. As with habitat restoration and averted loss offsets, biodiversity markets should only be implemented where the costs and risks associated with ecological equivalency can be adequately balanced.

2.2 Ecological Equivalency

The planning of biodiversity offsets should be based on equivalence, with biodiversity losses comparable to biodiversity gains, thus ensuring no net loss (Noga, 2014; Rosa, Novachi, & Sánchez, 2016). Yet a key component of biodiversity offsets is the fact that some of biodiversity will be lost (Gardner et al., 2013; Maron et al., 2016; Noga, 2014; Rosa et al., 2016). As such, a particular challenge for designing biodiversity offsets is ensuring that the expected loss is acceptable.

In terms of ecological equivalence, while “like-for-like” represents exact, or as near to, equivalence, trading up can be advantageous in some circumstances (Gardner et al., 2013). Trading up occurs where biodiversity offsets are steered to priority areas for both ecological and socio-economic investment in contrast to the requirement for the replacement of impacted resources in similar sites and in close proximity to the impacts (Tallis, Kennedy, Ruckelshaus, Goldstein & Kiesecker, 2015). Trading up can provide environmental benefits that are more valuable to developers, regulators and/or communities (Bull, Gordon, Watson & Maron, 2016; Takacs, 2018), and results in significant cost savings (Habib, Farr, Schneider & Boutin, 2013). Requirements for equivalent vegetation to be protected are up to two orders of magnitude greater in terms of area (Habib et al., 2013), requiring a significant burden not only on developers, but also on governments in regard to monitoring and assessment for compliance. Trading up can allow conservation to be focused on regional priorities (Habib et al., 2013). For example, Kujala, Whitehead, Morris and Wintle (2015) reported that biodiversity offsets that were developed to address strategic priorities led to a 10% increase in biodiversity, while like-for-like biodiversity offsets led to a 10% decrease in biodiversity. Trading up can provide additional compensation to areas that have experienced cumulative impacts. It is also valuable for practical purposes, as it can allow a broader range of offset locations (Habib et al., 2013), which is particularly important in areas where availability of land for biodiversity offsets is difficult due to tenure issues (Abdo et al. *in prep.*). In areas where there are biodiversity offsets markets, trading up can also facilitate market activity (Habib et al., 2013).

Trading up, unlike ‘like-for-like’ biodiversity offsets, can, however, remove visibility of the links between losses at the development site and gains at the offset site (Bull et al., 2016). As with other types of biodiversity offsets, whilst considering trading up it is important to ensure that loss at the development site and gains derived from conservation activities at the offset site are equivalent (Habib et al., 2013). It is also important to ensure that ecosystems with attributes that are less socially/politically desirable, but that provide a supportive or functional advantage for desirable ecosystems/species (e.g. areas that support key life stages of desirable species), are not excluded.

Non-equivalent biodiversity offsets should not be permitted unless in combination with trading up (Villarroya, Barros & Kiesecker, 2014). The determination of ecological equivalence should incorporate natural values, including consideration of biodiversity representation and species persistence (Andrello, Jacobi, Manel, Thuiller & Mouillot, 2015). Species persistence is strongly related to dispersal through population persistence, mean time to extinction, number of occupied habitat patches and metapopulation capacity (Andrello et al., 2015). Consideration of ecosystem components that will not be measured (e.g. habitat structure) and those aspects of

biodiversity that are important to communities but that do not necessarily provide a substantial conservation outcome (e.g. cultural values) (Gardner et al., 2013) are also important to ensure that biodiversity offsets do not create or deepen social inequities (Mandle et al., 2016; Rosa et al., 2016; Tallis et al., 2015) and can balance the principles of sustainable development.

2.3 Biodiversity Indicators as Representatives of Biodiversity

Ecological communities are unique, so it is impossible to exactly replace the biodiversity of one area in another, which from a practical sense, would be prohibitively costly and time consuming. As such, surrogates, proxies or indicators are chosen to represent aspects of biodiversity (Bezombes, Gaucherand, Spiegelberger, Gouraud & Kerbiriou, 2018; Duelli & Obrist, 2003; Kiesecker et al., 2009; Macintosh, 2015), particularly where there is a paucity of data available regarding the components, structure and/or function of the affected ecosystem (McElwee, 2017). Indicators are important basis of biodiversity offset markets as they contribute to the 'currency' that can be traded (Benabou, 2014).

Indicators are typically chosen to represent those aspects of an ecosystem that are the most important to communities, governments, developers and other relevant stakeholders (Coralie, Guillaume & Claude, 2015). As different components of ecosystems are valued by different stakeholders (Gardner et al., 2013), the choice of indicators should include stakeholder input and consideration of equity (Noga, 2014). This requires prioritization of rules and natural values (Macintosh, 2015). In practice, however, this can be difficult to define (Maseyk et al., 2016), as stakeholders can have competing priorities.

Several biodiversity indicators are required to ensure that desirable natural values are adequately represented. While directing conservation activities at a single indicator species would enable biodiversity offset gain calculations to be simplistic and cost effective, this approach to biodiversity offset would result in misrepresentation of natural values, increased variability in offset outcomes and lower resilience in the resultant ecosystem, potentially limiting the offset success (Duelli & Obrist, 2003; Ruppert, Hogg & Poesch, 2018). For example, despite undertaking conservation activities, indicator species may fail to thrive if conservation activities do not encompass other species the indicator species is clearly linked to. Additionally, the use of a single indicator species could create a false positive of success if the indicator species used responds positively to conservation actions, whilst other aspects of the impacted ecosystem either do not exist within in the designated biodiversity offset area or fail to thrive.

Each aspect of the ecosystem (or each aspect to be offset) requires a corresponding indicator (Duelli & Obrist, 2003; Qué-tier & Lavorel, 2011). Indicators should consider threatened and priority species, key species that are very specific to particular habitats, species with restrictive life histories, those that have lost significant habitat due to cumulative effects, and species that are particularly sensitive to human influence (Kiesecker et al., 2009). In addition to carefully selected indicator species, carbon, water and other indicators of condition (soil, vegetation) should also be included (King & Wilson, 2015). Genetic diversity is another important attribute, as sites may appear similar but have a different genetic composition, particularly in terms of less obvious components (e.g. microbes), that are essential to ecosystem success (Tierney, Sommerville, Tierney, Fatemi & Gross, 2017). While habitat type needs should be equivalent, greater gains in species richness may be achieved in areas with less remnant vegetation (Gibbons et al., 2018). As such, consideration of indicators should not rely solely on vegetation type and condition (Kujala et al., 2015) and should also incorporate structural, compositional and functional attributes (Rohr, Bernhardt, Cadotte & Clements, 2018). The biodiversity aspects of ecosystems and landscapes should be captured, as these in turn contribute to ecosystem function, and have societal benefits (Walz, 2015). A balance between rare and threatened species, and ecosystem functions and services is also required (Rohr et al., 2018). Biodiversity offsets also need to take into account external threats, such as natural disasters and climate change (May, Hobbs, & Valentine, 2017), as these threats have the ability to prevent the offset from reaching its' objectives.

Determining appropriate indicators is extremely important, as in their absence, concealed trades may occur. Concealed trades are exchanges of biodiversity elements that are not explicitly accounted for and which are either offset implicitly or lost in the exchange (e.g. different canopy tree species within the same vegetation type, or genes within species) (Maseyk et al., 2016). In order to avoid concealed trades, rigorous science must be applied to ensure that all natural values are known and that appropriate indicators for each natural value are included (Duelli & Obrist, 2003).

Indicators provide practical and cost advantages over attempts at compensation for all natural values of an ecosystem (Bennun, 2014); however, in order to be effective, indicators must be based on a grounding of robust science that has established links to the natural values of the ecosystem. Appropriate indicators that are both

representative and sensitive to changes from impact and conservation actions may overcome deficiencies in metrics used to determine no net loss (Bezombes et al., 2018). However, the assessment and monitoring of indicators must occur at a scale that is appropriate to each indicator, as scales such as those used by planning frameworks are often too broad to address the needs of individual species (Kormos, 2015).

3. Scale of Offset

Determining the size of biodiversity offsets relies on five key features: 1) definition of key species/ecosystems, 2) appropriate indicators, 3) calculation of the loss/gain, 4) understanding of time-lags, and 5) understanding of uncertainties and risks (Jacob, Vaissiere, Bas & Calvet, 2016). The size and the extent of the biodiversity offset must be adequate to compensate for relevant natural values, in order to ensure that biodiversity offsets adequately consider all aspects of sustainable development.

3.1 Size of Offset

The size of the biodiversity offset should be proportional to the size and scale of the environmental and social impact, and should incorporate the risk of failure (Fallding, 2014; Quétier & Lavorel, 2011; ten Kate, Bishop, & Bayon, 2004). Biodiversity offsets are often scaled in order to ensure adequate compensation for losses of type of compensatory measure, degree of ecosystem impact, time and space (Benabou, 2014). In terms of biodiversity offsets, the ratios used for this scaling is termed as “multipliers”. While multipliers represent the ratio between the offset area and the impacted area, they are usually much greater than one as they are used to compensate for deficiencies in offsetting (Moilanen & Laitila, 2015). Multipliers are often used to account for uncertainties in project design and implementation (Bull & Brownlie, 2015; Bull et al., 2016; Bull, Lloyd, & Strange, 2017b; Moilanen, van Teeffelen, Ben-Haim & Ferrier, 2009). Multipliers can also contribute to conservation objectives (Bull et al., 2017b) and reduce the risk of offset failure (Clarke & Bradford, 2014; McKenney, 2005; Quétier & Lavorel, 2011) by compensating for lack of information, imperfect exchanges or risk of failure (Bull et al., 2017b). They can also account for time lags (Bull et al., 2016; Gardner et al., 2013; Moilanen & Laitila, 2015; Moilanen et al., 2009) and conservation actions of a shorter duration than the environmental impact from development (Moilanen & Laitila, 2015). Requirements for multipliers can influence developers to provide impetus to avoid ecologically and/or socially important habitat (McKenney, 2005) through higher costs to both developers and regulators.

Multipliers should be developed to address residual risk after mitigation measures (Gardner et al., 2013) and include consideration of additionality, risk of failure and timeframes for achievement of milestones (McKenney & Kiesecker, 2010). Generally, however, biodiversity offset ratios either do not consider the ecosystem as a whole, climatic conditions or ongoing threats to the offset area (May et al., 2017), or are too small to adequately account for these attributes (Bell, 2016; Bull et al., 2017b). While multipliers are rarely required to be greater than a ratio of 1:10, they would often need to be in the tens to hundreds to truly achieve no net loss (Bull, Abatayo & Strange, 2017a). For example, Fallding (2014) reported that offset multipliers used in Australia ranged from 2:1 for key fish habitat offsets in NSW and certain vegetation offsets in Queensland, to 10:1 for wetland offsets in NSW and certain Commonwealth biodiversity offsets. Additionally, social considerations, such as social, ethical and governance concerns, are rarely addressed, which could result in the need for even larger multipliers (Bull et al., 2017a; Bull et al., 2017b). In order to be effective and to minimise these impacts, multipliers should be linked to risk and cost/benefit. They should ensure appropriate consideration of all aspects of sustainable development by incorporating biodiversity and socio-economic aspects, as well as future considerations (e.g. climate and ongoing threats).

While the size of the biodiversity offset will be proportional to the cost and risk of the offset, the definition of ecosystem extent (including land cover, land use, habitat) is also a key factor in determining the size of the offset. While this can be difficult and costly to identify, technology, such as satellite remote sensing, can be used to provide efficiencies (King & Wilson, 2015).

3.2 Consideration of Offset Extent

Policy objectives for biodiversity offsets and no net loss are typically at the site level (Bull et al., 2013; Burgin, 2008); however, this could lead to the uneven distribution of natural values (Budiharta et al., 2018). While it is generally accepted that biodiversity offsets should be selected at the smallest size at which conservation goals can be met (i.e. where no net loss is achieved) (Kiesecker et al., 2009), studies such as that by Di Minin et al. (2017) reported that small increases in targets at no additional costs can improve the representation of biodiversity and ecosystem services. As many biodiversity processes operate over larger scales (Fallding, 2014) and outside factors have the ability to impact on the success of the offset area (e.g. invasive species), biodiversity offsets should be considered at the landscape level (McKenney & Kiesecker, 2010; Noga, 2014). Consideration

of biodiversity offsets at a landscape level provides assurance of a number of key factors: that the environmental, social and economic significance of the area is accounted for; that no go areas and the most appropriate location(s) and suite of offset activities have been determined; and that future risks to the successful achievement of biodiversity offset goals have been identified (Gardner et al., 2013). These factors do not necessarily have to be addressed on an individual project basis, rather much of this information could be available through a strategic landscape-scale planning framework (henceforth 'planning framework').

Consideration of offsets at a landscape scale can create social issues because considering biodiversity offsets over a broader area involves a greater range of stakeholders, which may have differing views (Budiharta et al., 2018). Additionally, while increased connectivity has obvious ecosystem advantages (e.g. dispersal, migration), connectivity could create disadvantages, especially in areas where there is increased risk of disease outbreaks (Kormos, 2015), fire or susceptibility to climate change. As such, social considerations and risk mitigation are also important considerations for planning frameworks, particularly if used in conjunction with biodiversity offsets.

4. Location of Offset

The determination of an appropriate distance between the biodiversity offset and site of impact is subjective and depends on connectivity of landscape, range and dispersal of key species, supply/redundancy of ecosystem functions and services, availability of land, external pressures and maximum benefit of desirable features for both communities and regulators. The biodiversity offset should be located such that it provides the same desirable features as the development site (Quétier & Lavorel, 2011) and should provide complementary aspects to other intact/protected areas within the landscape (Kujala et al., 2015). Additionally, impacts on communities should be considered to ensure that the siting of the offset does not create social inequities (Ali, Kennedy, Kiesecker & Geng, 2018; Griffiths, Bull, Baker & Milner-Gulland, 2018; Jacob et al., 2016), or introduce/exacerbate leakage (Noga, 2014). Leakage occurs when the offset activity does not stop environmental damage, but merely displaces impacts to another location (Moilanen & Laitila, 2015; Pascual et al., 2017). For example, carbon-rich peat-swamp forests in Indonesia, that are encouraged as offsets for their carbon capture abilities, have been found to support lower levels of species diversity and threatened species than other ecosystems (Moilanen & Laitila, 2015).

There is no defined appropriate distance for an offset site in relation to a development site (Kiesecker et al., 2009; Moilanen & Kotiaho, 2018), although biodiversity offsets that are located near the development site, or at least in the same bioregion, provide additional biodiversity and social advantages. Biodiversity offsets that are further away from the development site, as is often the case with large-scale biodiversity offsets, can be less connected to and have less similar biodiversity than the development site (Yu, Cui, & Gibbons, 2018). While having the biodiversity offset close to where impact has occurred is usually preferred, in some circumstances (e.g. to reduce cost, for trading up or in order to improve habitat connectivity) it may be advantageous to locate the biodiversity offset elsewhere (Rogers & Burton, 2016; Rohr et al., 2018; Tallis et al., 2015). Requirements for biodiversity offsets to be sited as close to the development site as possible can hamper efforts to ensure it is part of a larger coordinated landscape scale plan (Lukey, Cumming, Paras, Kubiszewski & Lloyd, 2017), as well as hamper community access to ecosystem services (Bennun, 2014).

Biodiversity offsets that are not close the development site may require greater conservation efforts (e.g. higher multipliers) than those placed nearby. This is because natural environmental processes, such as dispersal and migration of species between the offset and the development site (or nearby sites within the bioregion), may not occur if the distance between the sites is too great (Yu et al., 2018). This lack of connectivity may increase the time taken for an offset to achieve its goals, and/or may increase the costs associated with maintenance and rehabilitation of the offset area. In order to balance the principles of sustainable development, the location of biodiversity offsets should be determined strategically as part of a landscape-scale planning framework that balances environmental, social and economic concerns for natural values.

5. Timescales Associated with Offsets

Time delays in the realisation of gains from a biodiversity offsets can be substantial, taking several decades to be realised, if ever (McKenney & Kiesecker, 2010; Moreno-Mateos et al., 2015). Time delays in the delivery of the gains from an offset may cause the loss of biodiversity and could also cause greater threat to certain species or even extinction (Gardner et al., 2013; Maron, Dunn, McAlpine & Apan, 2010). Additionally, these time delays can create issues in the provision of ecosystem services or intergenerational equity (King & Wilson, 2015; Overton, Stephens & Ferrier, 2013).

Different offset approaches may require consideration of different timescales to ensure environmental gains are

realised (Fallding, 2014). Flexibility in timing is often needed as, except in the case of advanced offsets, conservation actions related to biodiversity offsets are usually only initiated after development impacts have occurred (Moilanen & Kotiaho, 2018). While time lags in the realization of gains from biodiversity offsets might be permitted from a practical perspective, these should not in turn pose an imminent threat to the natural values requiring protection and/or compensation (Maron et al., 2012). Furthermore, they should not create inequities from political changes over time (Taherzadeh & Howley, 2016), with the biodiversity offset commitment lessening to the detriment of communities or increasing to the detriment of developers. The risk of time lags in the delivery of biodiversity offset outcomes can be lessened by having compensatory conservation activities start in advance of the development impact (advanced offsets) (Burgin, 2008; Gardner et al., 2013; Kiesecker et al., 2009; Lodhia, Martin, & Rice, 2018; Quétier & Lavorel, 2011; Villarroya et al., 2014). Alternately, the risks associated with time lags can be avoided or reduced by increasing the size/scale of the offset through use of a multiplier (Gardner et al., 2013; Villarroya et al., 2014), with longer time lags corresponding to offsets that are larger in scope and/or scale (Yu et al., 2018).

Time lags in the delivery of offset outcomes may be beneficial, as they can allow natural recovery rates to be observed (where these are not known prior to the implementation of the offset) (Rohr et al., 2018). However, habitats that take a long time to recover should not be compensated for by those that are easier to restore, as this could result in biodiversity loss (Gibbons et al., 2018), particularly where out of kind offsets are permitted (Overton et al., 2013). While time lags are generally unavoidable, the cost and risk to environmental, social and economic aspects should be balanced to ensure that the principles of sustainable development are maintained.

6. Duration of Offsets

Biodiversity offsets should focus on long-term conservation (Noga, 2014) and the environmental gains provided by biodiversity offsets should be permanent (Rosa et al., 2016). The impact from the development should be reversible to ensure that a loss of natural values does not occur (Bull et al., 2013), and/or that the conservation actions undertaken persist (Moilanen & Kotiaho, 2018). Although in some cases, such as where the goal of a biodiversity offset is to remove pests, this would mean that the biodiversity offsets would have to continue in perpetuity as reinvasion is likely to occur when conservation actions associated with the biodiversity offset cease (Norton & Warburton, 2014). While often desirable by regulators and communities, requirements for permanent maintenance of a biodiversity offset is not feasible in practice as future societal, political, and economic priorities cannot be effectively predicted at the inception of the offset (Norton & Warburton, 2014). As such, biodiversity offsets are often required to last as long as the impacts from development (Bull et al., 2013; Gardner et al., 2013; Githiru et al., 2015; Kiesecker et al., 2009; Simpson et al., 2017). Mechanisms to ensure longer-term management include financial instruments, such as a bond set aside as an endowment where the interest accrued from the funds covers the annual cost of management (Norton & Warburton, 2014).

In theory, biodiversity offset gains should last at least as long as residual effects of the development (Gardner et al 2013). In practice this is unlikely, based on the few available studies that have monitored biodiversity offset performance after implementation e.g. (Clare & Krogman, 2013; May et al., 2017; Taherzadeh & Howley, 2016). There are, however, two options in the case of uncertainty of biodiversity offset outcomes or where the achievement of biodiversity offset gains are unlikely to be achieved: i) either biodiversity offsets should not be permitted (especially in the case of rare, unique, endangered species or communities, or where species and communities hold unique cultural or economic value) or ii) biodiversity loss should be accepted with the knowledge that some replacement will occur. However, monitoring of the biodiversity offsets must occur to ensure that the biodiversity offset is effective in providing adequate compensation, and to ensure that no detrimental impacts to the environmental, social or economic aspects of natural values occur, violating the principles of sustainable development.

7. Monitoring and Measures of Offset Success

In order to assess the success (or failure) of an offset, there must be a set definition of what constitutes adequate/required compensation, as well as a timeframe for the delivery of this compensation (Brady & Boda, 2016). Biodiversity offset milestones should be measured against an appropriate baseline, ideally the counterfactual (Lindenmayer et al., 2017), as well as appropriate completion criteria (May et al., 2017). Completion criteria should be developed for all offset indicators so that appropriate measurements and monitoring of desirable aspects of the ecosystem are included (Takacs, 2018). For example, Lindenmayer et al. (2017) reported that despite attempts to reduce the impacts of developments on black cockatoos in Western Australia through installation of suitable nest boxes, the boxes were instead inhabited by exotic pest species, and thus while the offset requirements were considered 'completed', the original outcomes intended by

these offset requirements were not achieved. Monitoring is essential (Lindenmayer et al., 2017) and should include clear definitions, milestones, timeframes and monitoring methodologies (Fitzsimons & Carr, 2014; Koh, Hahn & Ituarte-Lima, 2014). Multiple aspects of the values to be offset should be measured to accurately determine offset performance (Maron et al., 2012) and to allow adaptive management measures to be undertaken. As suggested by Rohr et al. (2018), milestones should include genetic composition of species of interest, species' abundance, community composition, and ecosystem function.

Ecosystem function and services are important components of natural values, but there are no standard metrics or guidance on how these would be evaluated. McElwee (2017) suggests that the assessment of ecosystem production could be one way to assess these changes. This does not, however, provide an assessment of other cultural and community use/non-use values, which can only be assessed indirectly through community consultation. As such, biodiversity offset monitoring programs should assess ecological performance along with social and governance performance (Gelcich, Vargas, Carreras, Castilla & Donlan, 2017). This already occurs in some jurisdictions, such as France, where the development of biodiversity offsets requires negotiation with relevant stakeholders to ensure their interests are considered (Guillet & Semal, 2018). Stakeholder input and/or community consultation is needed to ensure that issues around conservation activities are resolved (Irtie, 2015; Rohr et al., 2018; Taherzadeh & Howley, 2016). Milestones also may need to include aspect of disturbance, as some species will only recruit to disturbed or structurally modified ecosystems (Tierney et al., 2017). However, biodiversity offset milestones should not just focus on natural values but include consideration of ecological stability and resilience (Rohr et al., 2018). This could occur through ongoing monitoring as part of a planning framework.

Biodiversity offsets should be monitored, at a minimum, until they reach their intended goals, in order to ensure that environmental, social and economic impacts to natural values are not ongoing (Villarroya et al., 2014). Monitoring of the biodiversity offset should occur until there is confidence that gains from conservation activities are persistent, particularly in cases where the impact of the development is not reversible. Ongoing monitoring is not necessarily the responsibility of the developer and should be shared with communities and regulators as part of a planning framework.

8. Discussion

8.1 Developing of a Holistic Model for Biodiversity Offsets

This review reports that determining appropriate scope, scale, location, timing, duration and monitoring components is key to the development of a holistic model for effective biodiversity offsets. An overview of best practice recommendations based on these key components is provided in Table 1. While there are several ways these components can be implemented in order to ensure that biodiversity offsets area effectively contribute to sustainable development, consideration of each component should include the environmental, social and economic aspects of natural values. However, while the consideration of the above demonstrates best practice and provides a holistic model balancing all aspects of sustainable development, the cost and risk associated with these components must also be considered to ensure that biodiversity offsets will be feasible.

Table 1. Holistic model describing best practice recommendations for biodiversity offsets

Component	Best practice recommendations
Scope	Biodiversity offset markets providing habitat restoration conservation activities
	Trading up that ensures equivalency of natural values
	Indicators for key natural values based on robust science and incorporating stakeholder input
Scale	Consideration at a landscape-scale
	Size that reflects application of multipliers used to mitigate risk
Location	Biodiversity offsets should be placed strategically where benefits are maximised, but impacts to environmental, social and economic concerns are minimised
Timing	Time lags should be minimised as far as possible
Duration	Biodiversity offsets should provide benefits that persist as long as the impacts from development
Monitoring	Stakeholder consultation and persist with monitoring until the biodiversity offset reaches its intended goals
	Include appropriate milestones and completion criteria for all natural values and should inform adaptive management

8.2 Cost and Risk Management

Key to developing a holistic model for biodiversity offsets is consideration of cost and risk management. To effectively consider cost and risk, the design and ongoing management of biodiversity offsets should include cost benefit analyses identifying key species, and also consider the cost of management, as exemplified by Carwardine et al. (2014) for conservation planning in the Pilbara region of Western Australia. Biodiversity offsets should also account for contingency costs if their intended goals are not achieved. For example, if the cost of compensation is too high, then it can increase the risk of offset failure, with developers unable to meet the costs associated with adequate compensation, and/or regulators unable to ensure adequate monitoring and enforcement. This outcome can have consequences for communities by way of taxation and/or reduced financing for other services. Ultimately, this will result in a loss of natural values.

Having transparent milestones and completion criteria can mitigate both the costs and risks of offset failure, as uncertainty can affect the viability of a development (Miller et al., 2015). As a consequence, this may have negative economic implications and result in a reduction of services available to communities. Offset failure can also be mitigated by integrating conservation actions within planning frameworks, ensuring that the offset area effectively compensates for the loss of natural values—and ensuring that reporting on offset outcomes is open and transparent (Koh et al., 2014). Similarly, the risk of offset failure can be mitigated by ensuring that adequate financing to ensure intended offset outcomes is in place prior to environmental impact (Brown & Penelope, 2016; Pilgrim et al., 2013), and by using bonds to cover costs in the event of failure and/or fines if offsets fail to reach pre-agreed milestones (Clarke & Bradford, 2014). The effectiveness of biodiversity offsets is also reliant on the availability of comprehensive and reliable datasets (Bull et al., 2018). Where this is not available, it is common for regulators to invoke the precautionary principle, allowing development to proceed, but requiring biodiversity offsets to incorporate additional measures of certainty, such as increased multipliers and/or bonds.

8.2.1 Inclusion of Bonds

The likelihood that an offset will succeed is usually based on environmental factors, but at the neglect of public support, community benefits and cost effectiveness (Noga, 2014). However, the social risks associated with uncertain offset outcomes can be mitigated if the public interest is protected in the event that the offset fails (Brown & Penelope, 2016). A key way to achieve this is through the requirement for a bond. A bond is a monetary sum that is held as insurance until a biodiversity offset achieves certain outcomes. These outcomes should include appropriate milestones and completion criteria. Bonds not only insure against non-delivery, but can also ensure duration of conservation outcomes (Norton & Warburton, 2014). Bonds are already required in some jurisdictions, such as the New South Wales Department of Primary Industries, which operates a policy of

'no net loss' for marine developments and requires a monetary bond to insure against failure (Burgin, 2008).

In conjunction with bonds, conservation covenants could be used to set aside areas of land for biodiversity offsets that are only used in the event that a development has an ecological impact. This could be particularly helpful for developments that occur in areas where there is a paucity of information on natural values. Where multipliers are used to compensate for risk of failure as opposed to a bond and/or covenant, the multiplier would be much higher, as observed by Moilanen et al. (2009). This results in further cost implications both for developers and regulators. While the use of a bond and/or covenant in this case could allow the developer to avoid high costs, time delays may mean that the eventual offset may need to be much larger than originally proposed to compensate for the delays. Given that multipliers required under the precautionary principle are particularly conservative, on balance this could work in the favour of the developer.

8.2.2 Incorporation of Biodiversity Offsets in Planning Frameworks

Monitoring and measurement can help biodiversity offsets meet their intended goals, but they will only be as effective as their metrics, which should be suitable and robust. While ecosystem indicators are developed to measure natural values, they can also be used to evaluate the performance of biodiversity offset-related conservation activities in meeting their intended goals (Bezombes et al., 2018). Determining appropriate goals for biodiversity offsets is, however, difficult, especially prior to the commencement of conservation measures (van Teeffelen et al., 2014). As such, biodiversity offsets should be implemented as part of a planning framework to ensure that milestones, goals and completion requirements for biodiversity offsets are implemented in a strategic and transparent way. A strategic planning framework including the environmental, social and economic aspects of all natural values can reduce risks to sustainable development.

Planning frameworks are key to ensuring that biodiversity offsets are delivered strategically, and to the greatest environmental, social and economic benefit. Frameworks can assist in the identification of suitable sites for offsets (Brownlie & Botha, 2009) by identifying areas not suitable for biodiversity offsets. Unsuitable areas would include locations with competing land tenure considerations or other issues that might ultimately hamper the success of the offset (e.g. anthropogenic pressures, such as fishing), or areas where risk is too high/costs are too great for compensation. While the use of biodiversity offsets markets can be used to fill gaps in conservation priorities (Iritie, 2015), this can also be achieved by ensuring that offsets are linked to planning frameworks. Ultimately, if used in combination, greater cost and risk efficiencies may be achieved.

Planning frameworks should only be implemented within defined project constraints (Macintosh, 2015). Defining these project boundaries is essential, especially when used in combination with offset markets, as offset markets can create financial incentives that influence the decision-making process in a way that is unfavourable to the conservation of biodiversity (Maron et al., 2016). This is particularly true of the negotiated approaches. The levy approach may operate in reverse to this, creating an incentive for regulators to seek funding for their 'conservation wish-list' rather than for projects that would compensate for the development. In this situation, developers are required to deliver projects that are in the remit of governments (Tahezadeh & Howley, 2016), meaning that social priorities may not be compensated. The financial compensation sought in this case might again be of a scale disproportionate to the impact, and could create inequities between developers, particularly if regulation allows high levels of flexibility. While planned contributions in legislation can be beneficial in avoiding these undesirable outcomes, legislation needs to be very prescriptive, which makes it difficult to ensure that all natural values of each unique ecosystem are considered. In order to capitalise on the obvious advantages of biodiversity offset markets, whilst avoiding the aforementioned failings, markets should provide adequate compensation for environmental, social and economic aspects through habitat restoration and/or averted loss offsets, and be administered by an independent organisation.

Planning frameworks that include adaptive management and contingency planning can ensure that biodiversity offsets are more effective (May et al., 2017), that conservation outcomes are enhanced (Koh et al., 2014; Underwood, 2011) and can provide greater environmental benefits at a lower cost than those implemented as stand-alone projects (Lukey et al., 2017). Frameworks can also identify opportunities for trading up (Tallis et al., 2015) and ensure that landscape connectivity is maintained, thereby promoting population persistence (Andrello et al., 2015) and improving the likelihood that a biodiversity offset will achieve its intended goals (Birkeland & Knight-Lenihan, 2016; Simpson et al., 2017). Integration of biodiversity offsets into planning frameworks may also reduce the risks associated with averted loss offsets (Moilanen & Laitila, 2015). Incorporation into planning frameworks could ensure more comprehensive assessments of losses and gains are undertaken by having this achieved external to developers that have an impetus to minimise time and costs (Benabou, 2014).

Planning frameworks can also insure against cumulative effects of smaller development projects. While

biodiversity offsets are often focussed only on larger infrastructure projects, as they generate more public concern, smaller projects in aggregation can be just as detrimental, if not more so (Guillet & Semal, 2018; Peel & Godden, 2005). Conversely, planning frameworks can also enable biodiversity offsets to be delivered as a series of smaller, interconnected sites, as opposed to one larger area. Small-scale offsets are acknowledged as difficult to implement (Fallding, 2014), are subject to an increased pressure from edge effects, have increased administrative and compliance costs and pose a risk of having a lesser environmental value than more connected and integrated areas (Lukey et al., 2017). However, when implemented strategically as part of a planning framework, small scale offsets can reduce risk of offset failure. Implementation in this way can mitigate the risks associated with offsets within one large area, such as lack of ecosystem response to conservation actions (Moilanen et al., 2009), ecosystem decline from uncontrollable external influences (e.g. natural disasters), or requirements for further development. The likelihood of offset success can be improved in small scaled projects, by i) having several smaller varied offset areas as part of a package, ii) incorporating areas requiring different conservation actions at spatially dispersed sites, and iii) ensuring that the effects of conservation are not reduced overall through edge effects or reduced habitat connectivity.

Yet the use of interconnected, smaller offset areas may pose associated social inequities. As such, planning frameworks should be developed through a public process. This process should include contribution from relevant experts and members of the community to ensure that aspects of the environment and all associated social concerns are represented. Involvement of stakeholders can reduce the risk of offset failure, particularly in terms of ensuring long-lasting offset gains (Koh et al., 2014). By consulting stakeholders on environmental and social priorities, it is more likely that biodiversity offsets can meaningfully contribute to sustainable development.

In Australia, planning frameworks, termed bioregional plans, are possible under Section 176 of the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). These plans are developed by the Minister for the Environment and include consideration of environmental and social aspects. While bioregional plans have been developed for marine areas, to date no plans have been developed for terrestrial areas. The marine bioregional plans are in themselves very broad, with only five plans to cover the entire Australian maritime area. Further, the EPBC Act has restricted its consideration of the environment to Matters of National Environmental Significance (MNES), so these plans have been developed to only consider MNES. While the Minister must ensure public consultation of the draft plan, social considerations are not directly addressed by the plans. Finally, the plans are not regarded as a legislative instrument, but rather provide further information for the Minister's consideration when making a determination. In order to satisfactorily ensure the strategic use of biodiversity in Australia to contribute to sustainable development, a planning framework is needed that can be used as a legislative instrument and ensures more detailed plans encompassing consideration of all environment, social and economic aspects.

In the absence of planning frameworks, the location and intended conservation activities for a biodiversity offset must meaningfully contribute to sustainable development. This could involve a type of sustainable impact assessment (SIA) conducted on the offset itself, either as part of the development of an environmental impact assessment, or independently after conditional development approval that requires biodiversity offsets. An effective SIA involving stakeholder consultation will not only identify potential negative environmental, social and economic impacts, but will also assist in gaining social license to operate, reducing risks associated with offset failure. A SIA will also provide cost efficiencies in terms of monitoring, compliance and enforcement, through clearly identifying relevant key performance indicators and completion criteria.

8.2.3 Use of Advanced Offsets in the Planning Framework

Advanced offsets are those that have been implemented prior to development and have reached their intended goals (Abdo et al. 2019). While the concept of advanced offsets is recognised by many regulators to be effective, this approach requires significant strategic planning (Bell, 2016), highlighting the need for a planning framework. In this way, planning frameworks could enable a bank of conservation sites that are delivered by several different parties, such as land-holders, government agencies or non-government conservation organisations that could then be used by developers as appropriate at a later date. Developers would provide a monetary sum (fee) for that biodiversity offset. Planning frameworks link with biodiversity offset markets in a strategic way and provide a biodiversity 'savings bank', ensuring there is a continuous overall net gain of biodiversity that increases with newer conservation projects reaching their goals and diminishes with deleterious impacts from development.

Implementation of advanced offsets as part of a planning framework would also prevent duration issues, as the biodiversity offset would have, by definition, already achieved its goals. Costs for developers would be known,

as associated conservation activities would have already been undertaken. Furthermore, if the advanced offset was 'certified' as having achieved its intended outcomes, costs for regulators (and therefore communities) would also be lower as ongoing monitoring and enforcement would not be required. Identification through a planning framework of suitable sites and conservation activities for biodiversity offsets would also allow flexibility for developers to provide advanced offsets where they have overcompensated and/or developed conservation programs under the planning framework. These offsets could be traded with other developers, employing 'peer-to-peer trading' such as has been proposed for energy providers (for example RENEW Nexus peer to peer energy trading project in Western Australia). This would not only provide efficiencies and reduce risk in terms of biodiversity benefits but would also reduce costs as the price of the advanced offset would be set by demand and not forecast on anticipated conservation activities potentially subject to change depending on future environmental, social and political needs.

While there are currently several different methods used for conservation planning available (e.g. Kiesecker et al., 2009), these are rarely suitable for the identification of areas for biodiversity offsets as they do not account for offset-specific factors such as additionality and equivalence (Yu et al., 2018). As such, these methods must be used in combination with other techniques.

9. Conclusion

This manuscript presents a holistic model for the design, implementation and ongoing management of direct biodiversity offsets incorporating all aspects (environmental, social, economic) of sustainable development. This holistic approach to biodiversity offsets is imperative to ensure that biodiversity offsets meaningfully contribute to sustainable development and to prevent loss of natural values and/or creation of socioeconomic inequities. While this holistic approach could be applied through a SIA, a more efficient and potentially more effective approach would have biodiversity offsets being considered strategically as part of a planning framework. In order to minimise costs and risk, thus ensuring optimal efficiency, planning frameworks used by biodiversity offsets should identify strategic opportunities for trading up, advanced offsets and adaptive management, as well as provide assurance of no net loss through the use of multipliers and/or bonds. Planning frameworks should encourage and support biodiversity offset markets and/or peer to peer trading to provide further cost saving efficiencies for both developers and regulators (and by default, communities). This should occur simultaneously with managing the risk of creating social inequities. In this way, biodiversity offsets would be able to tangibly contribute to the Sustainable Development Goals of each jurisdiction, ensuring access to new resources that are not to the detriment of the environment or communities.

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References

- Abdo, L., Griffin, S., & Kemp, A. (2019). Apples for Oranges: Disparities in Offset Legislation and Policy among Jurisdictions and its Implications for Environmental Protection and Sustainable Development in Australia. *Environmental Management and Sustainable Development*, 8(1). <https://doi.org/10.5296/emsd.v8i1.14081>
- Abdo, L., Griffin, S., Kemp, A., & Coupland, G. (in prep.) Disparity in biodiversity offset regulation across Australia may impact their effectiveness.
- Ali, M., Kennedy, C. M., Kiesecker, J., & Geng, Y. (2018). Integrating biodiversity offsets within Circular Economy policy in China. *Journal of Cleaner Production*, 185, 32-43. <https://doi.org/10.1016/j.jclepro.2018.03.027>
- Andrello, M., Jacobi, M. N., Manel, S., Thuiller, W., & Mouillot, D. (2015). Extending networks of protected areas to optimize connectivity and population growth rate. *Ecography*, 38(3), 273-282. <https://doi.org/10.1111/ecog.00975>
- Bekessy, S. A., Wintle, B. A., Lindenmayer, D. B., McCarthy, M. A., Colyvan, M., Burgman, M. A., & Possingham, H. P. (2010). The biodiversity bank cannot be a lending bank. *Conservation Letters*, 3, 151-158. <https://doi.org/10.1111/j.1755-263X.2010.00110.x>

- Bell, J. (2016). Implementing an outcomes-based approach to marine biodiversity offsets: lessons from the Great Barrier Reef. *Australasian Journal of Environmental Management*, 1-16. <https://doi.org/10.1080/14486563.2015.1081837>
- Benabou, S. (2014). Making Up for Lost Nature? A Critical Review of the International Development of Voluntary Biodiversity Offsets. *Environment and Society*, 5(1). <https://doi.org/10.3167/ares.2014.050107>
- Bennun, L. A., Ekstrom, J., & Bull, J. (2014). *Integrating the value of natural capital into private and public investment: the role of information*. The Biodiversity Conservancy, Cambridge, U.K.
- Bezombes, L., Gaucherand, S., Spiegelberger, T., Gouraud, V., & Kerbiriou, C. (2018). A set of organized indicators to conciliate scientific knowledge, offset policies requirements and operational constraints in the context of biodiversity offsets. *Ecological Indicators*, 93, 1244-1252. <https://doi.org/10.1016/j.ecolind.2018.06.027>
- Bidaud, C., Schreckenber, K., & Jones, J. P. G. (2018). The local costs of biodiversity offsets: Comparing standards, policy and practice. *Land Use Policy*, 77, 43-50. <https://doi.org/10.1016/j.landusepol.2018.05.003>
- Bigard, C., Pioch, S., & Thompson, J. D. (2017). The inclusion of biodiversity in environmental impact assessment: Policy-related progress limited by gaps and semantic confusion. *Journal of Environmental Management*, 200, 35-45. <https://doi.org/10.1016/j.jenvman.2017.05.057>
- Birkeland, J., & Knight-Lenihan, S. (2016). Biodiversity offsetting and net positive design. *Journal of Urban Design*, 21(1), 50-66. <https://doi.org/10.1080/13574809.2015.1129891>
- Brady, A. F., & Boda, C. S. (2016). How do we know if managed realignment for coastal habitat compensation is successful? Insights from the implementation of the EU Birds and Habitats Directive in England. *Ocean & Coastal Management*. <https://doi.org/10.1016/j.ocecoaman.2016.11.013>
- Brown, M., A., & Penelope, J. (2016). Biodiversity offsets in New Zealand: addressing the risks and maximising the benefits. *Policy Quarterly*, 12(1), 35-41. <https://doi.org/10.26686/pq.v12i1.4580>
- Brownlie, S., & Botha, M. (2009). Biodiversity offsets: adding to the conservation estate, or 'no net loss'? *Impact Assessment and Project Appraisal*, 27(3), 227-231. <https://doi.org/10.3152/146155109X465968>
- Budiharta, S., Meijaard, E., Gaveau, D. L. A., Struebig, M. J., Wilting, A., Kramer-Schadt, S., ... Wilson, K. A. (2018). Restoration to offset the impacts of developments at a landscape scale reveals opportunities, challenges and tough choices. *Global Environmental Change*, 52, 152-161. <https://doi.org/10.1016/j.gloenvcha.2018.07.008>
- Bull, J. W., & Brownlie, S. (2015). The transition from No Net Loss to a Net Gain of biodiversity is far from trivial. *Oryx*, 1-7. <https://doi.org/10.1017/S0030605315000861>
- Bull, J. W., Abatayo, A. L., & Strange, N. (2017a). Counterintuitive proposals for trans-boundary ecological compensation under 'no net loss' biodiversity policy. *Ecological Economics*, 142, 185-193. <https://doi.org/10.1016/j.ecolecon.2017.06.010>
- Bull, J. W., Brauneder, K., Darbi, M., Van Teeffelen, A. J. A., Quétier, F., Brooks, S. E., Dunnett, S., & Strange, N. (2018). Data transparency regarding the implementation of European 'no net loss' biodiversity policies. *Biological Conservation*, 218, 64-72. <https://doi.org/10.1016/j.biocon.2017.12.002>
- Bull, J. W., Gordon, A., Watson, J. E. M., & Maron, M. (2016). Seeking convergence on the key concepts in 'no net loss' policy. *Journal of Applied Ecology*. <https://doi.org/10.1111/1365-2664.12726>
- Bull, J. W., Hardy, M. J., Moilanen, A., & Gordon, A. (2015). Categories of flexibility in biodiversity offsetting, and their implications for conservation. *Biological Conservation*. <https://doi.org/10.1016/j.biocon.2015.08.003>
- Bull, J. W., Lloyd, S. P., & Strange, N. (2017). Implementation Gap between the Theory and Practice of Biodiversity Offset Multipliers. *Conservation Letters*, 10(6), 656-669. <https://doi.org/10.1111/conl.12335>
- Bull, J. W., Suttle, K. B., Gordon, A., Singh, N. J., & Milner-Gulland, E. J. (2013). Biodiversity offsets in theory and practice. *Oryx*, 47(03), 369-380. <https://doi.org/10.1017/S003060531200172X>
- Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F., & Rey-Benayas, J. M. (2011). Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology and Evolution* October, 26(10), 541-549. <https://doi.org/10.1016/j.tree.2011.06.011>

- Burgin, S. (2008). BioBanking: an environmental scientist's view of the role of biodiversity banking offsets in conservation. *Biodiversity and Conservation*, 17(4), 807-816. <https://doi.org/10.1007/s10531-008-9319-2>
- Business and Biodiversity Offsets Programme (BBOP). (2012). *Standard on Biodiversity Offsets*. BBOP, Washington, D.C. Retrieved March 13, 2019, from http://bbop.forest-trends.org/pages/biodiversity_offsets
- Business and Biodiversity Offsets Programme (BBOP). (2019). *Biodiversity Offsets*. BBOP, Washington, D.C. Retrieved March 13, 2019, from http://bbop.forest-trends.org/pages/biodiversity_offsets
- Carreras Gamarra, M. J., Lassoie, J. P., & Milder, J. (2018). Accounting for no net loss: A critical assessment of biodiversity offsetting metrics and methods. *Journal of Environmental Management*, 220, 36-43. <https://doi.org/10.1016/j.jenvman.2018.05.008>
- Carwardine, J., Nicol, S., Van Leeuwen, S., Walters, B., Firn, J., Reeson, A., . . . Chades, I. (2014). *Priority threat management for Pilbara species of conservation significance*. CSIRO, Brisbane, QLD.
- Clare, S., & Krogman, N. (2013). Bureaucratic slippage and environmental offset policies: the case of wetland management in Alberta. *Society & Natural Resources*, 26(6), 672-687. <https://doi.org/10.1080/08941920.2013.779341>
- Clarke, K. D., & Bradford, M. J. (2014). *A review of equivalency in offsetting policies*. Fisheries and Oceans Canada, Canadian Science Advisory Secretariat, Ottawa, Canada.
- Coggan, A., Buitelaar, E., Bennett, J., & Whitten, S. M. (2013). Transferable Mitigation of Environmental Impacts of Development: Two Cases of Offsets in Australia. *Journal of Environmental Policy & Planning*, 15(2), 303-322. <https://doi.org/10.1080/1523908X.2013.781350>
- Coralie, C., Guillaume, O., & Claude, N. (2015). Tracking the origins and development of biodiversity offsetting in academic research and its implications for conservation: A review. *Biological Conservation*. <https://doi.org/10.1016/j.biocon.2015.08.036>
- Department of Sustainability, Environment, Water, Population and Communities. (2012). *Environment Protection and Biodiversity Conservation Act 1999 Environmental Offsets Policy*. Department of Sustainability, Environment, Water, Population and Communities, Canberra, ACT.
- Di Minin, E., Soutullo, A., Bartesaghi, L., Rios, M., Szephegyi, M. N., & Moilanen, A. (2017). Integrating biodiversity, ecosystem services and socio-economic data to identify priority areas and landowners for conservation actions at the national scale. *Biological Conservation*, 206, 56-64. <https://doi.org/10.1016/j.biocon.2016.11.037>
- Duelli, P., & Obrist, M. K. (2003). Biodiversity indicators: The choice of values and measures. *Agriculture, Ecosystems and Environment*, 98(1), 87-98. [https://doi.org/10.1016/S0167-8809\(03\)00072-0](https://doi.org/10.1016/S0167-8809(03)00072-0)
- Fallding, M. (2014). Biodiversity offsets: Practice and promise. *Environmental and Planning Law Journal*, 31(11), 11-33.
- Filoché, G. (2017). Playing musical chairs with land use obligations: Market-based instruments and environmental public policies in Brazil. *Land Use Policy*, 63, 20-29. <https://doi.org/10.1016/j.landusepol.2017.01.012>
- Fitzsimons, J. A., & Carr, C. B. (2014). Conservation covenants on private land: issues with measuring and achieving biodiversity outcomes in Australia. *Environmental Management*, 54(3), 606-616. <https://doi.org/10.1007/s00267-014-0329-4>
- Gardner, T. A., Von Hase, A., Brownlie, S., Ekstrom, J. M., Pilgrim, J. D., Savy, C. E., . . . Ten Kate, K. (2013). Biodiversity offsets and the challenge of achieving no net loss. *Conservation Biology*, 27(6), 1254-1264. <https://doi.org/10.1111/cobi.12118>
- Gelcich, S., Vargas, C., Carreras, M. J., Castilla, J. C., & Donlan, C. J. (2017). Achieving biodiversity benefits with offsets: Research gaps, challenges, and needs. *Ambio*, 46(2), 184-189. <https://doi.org/10.1007/s13280-016-0810-9>
- Ghosh, S. (2015, April 18). Capitalisation of nature: Political economy of forest/biodiversity offsets. *Economic and Political Weekly*, L, 53-60.
- Ghosh, S. (2017). Compensatory Afforestation: 'Compensating' Loss of Forests or Disguising Forest Offsets? *Economic & Political Weekly*, LII(38), 67-75.
- Gibbons, P., Macintosh, A., Constable, A. L., & Hayashi, K. (2018). Outcomes from 10 years of biodiversity

- offsetting. *Global Change Biology*, 24(2), e643-e654. <https://doi.org/10.1111/gcb.13977>
- Gibson, R. B. (2009). Beyond the pillars: Sustainability assessment as a framework for effective integration of social, economic and ecological considerations in significant decision-making. In Sheate, W. R. (Ed.), *Tools, Techniques and Approaches for Sustainability: Collected Writings in Environmental Assessment Policy and Management*. World Scientific Publishing Co Pte Ltd., Singapore.
- Githiru, M., King, M. W., Bauche, P., Simon, C., Boles, J., Rindt, C., & Victurine, R. (2015). Should biodiversity offsets help finance underfunded Protected Areas? *Biological Conservation*, 191, 819-826. <https://doi.org/10.1016/j.biocon.2015.07.033>
- Godden, D., & Vernon, D. (2003). *Theoretical issues in using offsets for managing biodiversity*. Paper presented at the Annual Conference of the Australian Agricultural and Resource Economics Society, Fremantle, February.
- Gonçalves, B., Marques, A., Soares, A. M. V. D. M., & Pereira, H. M. (2015). Biodiversity offsets: from current challenges to harmonized metrics. *Current Opinion in Environmental Sustainability*, 14, 61-67. <https://doi.org/10.1016/j.cosust.2015.03.008>
- Griffiths, V. F., Bull, J. W., Baker, J., & Milner-Gulland, E. J. (2018). No net loss for people and biodiversity. *Conservation Biology*. <https://doi.org/10.1111/cobi.13184>
- Guillet, F., & Semal, L. (2018). Policy flaws of biodiversity offsetting as a conservation strategy. *Biological Conservation*, 221, 86-90. <https://doi.org/10.1016/j.biocon.2018.03.001>
- Habib, T. J., Farr, D. R., Schneider, R. R., & Boutin, S. (2013). Economic and ecological outcomes of flexible biodiversity offset systems. *Conservation Biology*, 27(6), 1313-1323. <https://doi.org/10.1111/cobi.12098>
- International Institute for Sustainable Development. (2017). *Topic: Sustainable development*. Retrieved from <http://www.iisd.org/topic/sustainable-development>
- Iritie, B. G. J. J. (2015). Economic Growth and Biodiversity: An Overview Conservation Policies in Africa. *Journal of Sustainable Development*, 8(2). <https://doi.org/10.5539/jsd.v8n2p196>
- Jacob, C., Buffard, A., Pioch, S., & Thorin, S. (2017). Marine ecosystem restoration and biodiversity offset. *Ecological Engineering*. <https://doi.org/10.1016/j.ecoleng.2017.09.007>
- Jacob, C., Vaissiere, A.-C., Bas, A., & Calvet, C. (2016). Investigating the inclusion of ecosystem services in biodiversity offsetting. *Ecosystem Services*, 21, 92-102. <https://doi.org/10.1016/j.ecoser.2016.07.010>
- Kiesecker, J. M., Copeland, H., Pocewicz, A., Nibbelink, N., McKenney, B., Dahlke, J., ... Stroud, D. (2009). A framework for implementing biodiversity offsets: selecting sites and determining scale. *BioScience*, 59(1), 77-84. <https://doi.org/10.1525/bio.2009.59.1.11>
- King, S., & Wilson, L. (2015). Experimental Biodiversity Accounting as a component of the System of Environmental- Economic Accounting Experimental Ecosystem Accounting (SEEA-EEA). Supporting document to the Advancing the SEEA Experimental Ecosystem Accounting project. United Nations.
- Kleining, B. (2017). Biodiversity protection under the habitats directive: Is habitats banking our new hope? *Environmental Law Review*, 19(2), 113-125. <https://doi.org/10.1177/1461452917714442>
- Koh, N. S., Hahn, T., & Ituarte-Lima, C. (2014). *A comparative analysis of ecological compensation programs: The effect of program design on the social and ecological outcomes*. Stockholm Resilience Centre, Sweden.
- Kormos, R., Mead, D., & Vinnedge, B. (2015). *Biodiversity offsetting in the United States: Lesson learned on maximising their ecological potential*. Retrieved from https://assets.fauna-flora.org/wp-content/uploads/2017/12/FFI_2015_Biodiversity-offsets-USA.pdf
- Kujala, H., Whitehead, A. L., Morris, W. K., & Wintle, B. A. (2015). Towards strategic offsetting of biodiversity loss using spatial prioritization concepts and tools: A case study on mining impacts in Australia. *Biological Conservation*. <https://doi.org/10.1016/j.biocon.2015.08.017>
- Lindenmayer, D. B., Crane, M., Evans, M. C., Maron, M., Gibbons, P., Bekessy, S., & Blanchard, W. (2017). The anatomy of a failed offset. *Biological Conservation*, 210, 286-292. <https://doi.org/10.1016/j.biocon.2017.04.022>
- Lodhia, S., Martin, N., & Rice, J. (2018). Appraising offsets as a tool for integrated environmental planning and management. *Journal of Cleaner Production*, 178, 34-44. <https://doi.org/10.1016/j.jclepro.2018.01.004>
- Lukey, P., Cumming, T., Paras, S., Kubiszewski, I., & Lloyd, S. (2017). Making biodiversity offsets work in

- South Africa – A governance perspective. *Ecosystem Services*, 27, 281-290. <https://doi.org/10.1016/j.ecoser.2017.05.001>
- Macintosh, A. (2015). The impact of ESD on Australia's environmental institutions. *Australasian Journal of Environmental Management*, 22(1), 33-45. <https://doi.org/10.1080/14486563.2014.999724>
- Mandle, L., Douglass, J., Lozano, J. S., Sharp, R. P., Vogl, A. L., Denu, D., . . . Tallis, H. (2016). OPAL: An open-source software tool for integrating biodiversity and ecosystem services into impact assessment and mitigation decisions. *Environmental Modelling & Software*, 84, 121-133. <https://doi.org/10.1016/j.envsoft.2016.06.008>
- Maron, M. (2012). Replacing lost ecosystems - the Devil is in the detail: Balancing biodiversity offsets with restoration reality. *Decision Point*, 63.
- Maron, M., & Louis, W. R. (2018). Does it matter why we do restoration? Volunteers, offset markets and the need for full disclosure. *Ecological Management & Restoration*, 19, 73-78. <https://doi.org/10.1111/emr.12330>
- Maron, M., Bull, J. W., Evans, M. C., & Gordon, A. (2015). Locking in loss: Baselines of decline in Australian biodiversity offset policies. *Biological Conservation*, 192, 504-512. <https://doi.org/10.1016/j.biocon.2015.05.017>
- Maron, M., Dunn, P. K., McAlpine, C. A., & Apan, A. (2010). Can offsets really compensate for habitat removal? The case of the endangered red-tailed black-cockatoo. *Journal of Applied Ecology*, 47, 348-355. <https://doi.org/10.1111/j.1365-2664.2010.01787.x>
- Maron, M., Hobbs, R. J., Moilanen, A., Matthews, J. W., Christie, K., Gardner, T. A., . . . McAlpine, C. A. (2012). Faustian bargains? Restoration realities in the context of biodiversity offset policies. *Biological Conservation*, 155, 141-148. <https://doi.org/10.1016/j.biocon.2012.06.003>
- Maron, M., Ives, C. D., Kujala, H., Bull, J. W., Maseyk, F. J. F., Bekessy, S., . . . Evans, M. C. (2016). Taming a Wicked Problem: Resolving Controversies in Biodiversity Offsetting. *BioScience*. <https://doi.org/10.1093/biosci/biw038>
- Maseyk, F. J. F., Barea, L. P., Stephens, R. T. T., Possingham, H. P., Dutson, G., & Maron, M. (2016). A disaggregated biodiversity offset accounting model to improve estimation of ecological equivalency and no net loss. *Biological Conservation*, 204, 322-332. <https://doi.org/10.1016/j.biocon.2016.10.016>
- May, J., Hobbs, R. J., & Valentine, L. E. (2017). Are offsets effective? An evaluation of recent environmental offsets in Western Australia. *Biological Conservation*, 206, 249-257. <https://doi.org/10.1016/j.biocon.2016.11.038>
- McDonald, J., McCormack, P. C., & Foerster, A. (2016). Promoting resilience to climate change in Australian Conservation Law: The case of Biodiversity Offsets. *UNSW Law Journal*, 39(4), 1612-1651.
- McElwee, P. (2017). The Metrics of Making Ecosystem Services. *Environment and Society*, 8(1). <https://doi.org/10.3167/ares.2017.080105>
- McKenney, B. (2005). *Environmental Offset Policies, Principles, and Methods: A Review of Selected Legislative Frameworks*. Biodiversity Neutral Initiative.
- McKenney, B. A., & Kiesecker, J. M. (2010). Policy development for biodiversity offsets: a review of offset frameworks. *Environmental Management*, 45(1), 165-176. <https://doi.org/10.1007/s00267-009-9396-3>
- Miller, K. L., Trezise, J. A., Kraus, S., Dripps, K., Evans, M. C., Gibbons, P., . . . Maron, M. (2015). The development of the Australian environmental offsets policy: from theory to practice. *Environmental Conservation*, 1-9. <https://doi.org/10.1017/S037689291400040X>
- Moilanen, A., & Kotiaho, J. S. (2018). Fifteen operationally important decisions in the planning of biodiversity offsets. *Biological Conservation*, 227, 112-120. <https://doi.org/10.1016/j.biocon.2018.09.002>
- Moilanen, A., & Laitila, J. (2015). Indirect leakage leads to a failure of avoided loss biodiversity offsetting. *Journal of Applied Ecology*, 53(1) 106-111. <https://doi.org/10.1111/1365-2664.12565>
- Moilanen, A., van Teeffelen, A. J. A., Ben-Haim, Y., & Ferrier, S. (2009). How Much Compensation is Enough? A Framework for Incorporating Uncertainty and Time Discounting When Calculating Offset Ratios for Impacted Habitat. *Restoration Ecology*, 17(4), 470-478. <https://doi.org/10.1111/j.1526-100X.2008.00382.x>
- Moldan, B. & Dahl, A.L. (2007). Challenges to Sustainability Indicators. In Hák, T., Moldan, B. & Dahl, A.L.

- (Eds). *Sustainability indicators: A scientific assessment*. Island Press, Washington DC.
- Moreno-Mateos, D., Maris, V., Béchet, A., & Curran, M. (2015). The true loss caused by biodiversity offsets. *Biological Conservation*. <https://doi.org/10.1016/j.biocon.2015.08.016>
- Nijnik, M., & Miller, D. (2017). Valuation of ecosystem services: paradox or Pandora's box for decision-makers? *One Ecosystem*, 2, e14808. <https://doi.org/10.3897/oneeco.2.e14808>
- Niner, H. J., Milligan, B., Jones, P. J. S., & Styan, C. A. (2017). Realising a vision of no net loss through marine biodiversity offsetting in Australia. *Ocean & Coastal Management*, 148, 22-30. <https://doi.org/10.1016/j.ocecoaman.2017.07.006>
- Noga, W. (2014). *Two papers on the cost effectiveness of conservation programs* (Masters), University of Alberta, Alberta, Canada.
- Norton, D. A., & Warburton, B. (2014). The Potential for Biodiversity Offsetting to Fund Effective Invasive Species Control. *Conservation Biology*. <https://doi.org/10.1111/cobi.12345>
- Overton, J. M., Stephens, R. T., & Ferrier, S. (2013). Net present biodiversity value and the design of biodiversity offsets. *Ambio*, 42(1), 100-110. <https://doi.org/10.1007/s13280-012-0342-x>
- Ozdemiroglu, E., Kriström, B., Cole, S., Riera, P., & Borrego, D. A. (2009). Environmental Liability Directive and the use of economics in compensation, offsets and habitat banking. *Proceedings of UK Network for Environmental Economists, London, England, March*.
- Pascual, U., Palomo, I., Adams, W. M., Chan, K. M. A., Daw, T. M., Garmendia, E., ... Phelps, J. (2017). Off-stage ecosystem service burdens: A blind spot for global sustainability. *Environmental Research Letters*, 12(7), 075001. <https://doi.org/10.1088/1748-9326/aa7392>
- Peel, J., & Godden, L. (2005). Australian environmental management: a 'dams' story. *University of New South Wales Law Journal*, 28(3), 668-695.
- Pilgrim, J. D., Brownlie, S., Ekstrom, J. M. M., Gardner, T. A., von Hase, A., ten Kate, K., ... Treweek, J. (2013). A process for assessing the offsetability of biodiversity impacts. *Conservation Letters*, 6(5), 376-384. <https://doi.org/10.1111/conl.12002>
- Quétiér, F., & Lavorel, S. (2011). Assessing ecological equivalence in biodiversity offset schemes: Key issues and solutions. *Biological Conservation*, 144(12), 2991-2999. <https://doi.org/10.1016/j.biocon.2011.09.002>
- Rogers, A. A., & Burton, M. P. (2016). *Public preferences for the design of biodiversity offset policies in Australia*. Working Paper 1601, School of Agricultural and Resource Economics, University of Western Australia, Crawley, Australia.
- Rohr, J. R., Bernhardt, E. S., Cadotte, M. W., & Clements, W. H. (2018). The ecology and economics of restoration: when, what, where, and how to restore ecosystems. *Ecology and Society*, 23(2). <https://doi.org/10.5751/ES-09876-230215>
- Rosa, J., Novachi, G., & Sánchez, L. E. (2016, May 11-14). *Offsetting and compensating biodiversity and ecosystem services losses in mining*. Paper presented at the IAIA16: 36th Annual Conference of the International Association for Impact Assessment, Nagoya Congress Center, Aichi-Nagoya, Japan.
- Ruppert, J. L. W., Hogg, J., & Poesch, M. S. (2018). Community assembly and the sustainability of habitat offsetting targets in the first compensation lake in the oil sands region in Alberta, Canada. *Biological Conservation*, 219, 138-146. <https://doi.org/10.1016/j.biocon.2018.01.014>
- Scholte, S. S. K., van Zanten, B. T., Verburg, P. H., & van Teeffelen, A. J. A. (2016). Willingness to offset? Residents' perspectives on compensating impacts from urban development through woodland restoration. *Land Use Policy*, 58, 403-414. <https://doi.org/10.1016/j.landusepol.2016.08.008>
- Simpson, K., de Vries, F. P., Armsworth, P., & Hanley, N. (2017). *Designing Markets for Biodiversity Offsets: Lessons from Tradable Pollution Permits*. Retrieved from <http://www.st-andrews.ac.uk/gsd/research/envecon/eediscus/>
- Taherzadeh, O., & Howley, P. (2016). *No net loss of what, for whom? Stakeholder perspectives on Biodiversity Offsetting in England*. Stockholm Environment Institute Working Paper 2016-11, Stockholm Environment Institute, Stockholm, Sweden
- Takacs, D. (2018). Are Koalas fungible? Biodiversity offsetting and the law. *N.Y.U. Environmental Law Journal*, 26(2), 161-226.

- Tallis, H., Kennedy, C. M., Ruckelshaus, M., Goldstein, J., & Kiesecker, J. M. (2015). Mitigation for one & all: An integrated framework for mitigation of development impacts on biodiversity and ecosystem services. *Environmental Impact Assessment Review*, 55, 21-34. <https://doi.org/10.1016/j.eiar.2015.06.005>
- ten Kate, K., Bishop, J., & Bayon, R. (2004). *Biodiversity offsets: Views, experience, and the business case*. IUCN, Gland, Switzerland and Cambridge, UK, and Insight Investment, London, UK.
- Tierney, D. A., Sommerville, K. D., Tierney, K. E., Fatemi, M., & Gross, C. L. (2017). Trading populations—can biodiversity offsets effectively compensate for population losses? *Biodiversity and Conservation*, 26(9), 2115-2131. <https://doi.org/10.1007/s10531-017-1348-2>
- Underwood, J. G. (2011). Combining landscape-level conservation planning and biodiversity offset programs: a case study. *Environmental Management*, 47(1), 121-129. <https://doi.org/10.1007/s00267-010-9589-9>
- United Nations. (2019). *Sustainable Development Goals: 17 goals to transform our world*. United Nations, Nairobi, Kenya. Retrieved March 29, 2019, from <https://www.un.org/sustainabledevelopment>
- van Teeffelen, A. J. A., Opdam, P., Wätzold, F., Hartig, F., Johst, K., Drechsler, M., ... Quétier, F. (2014). Ecological and economic conditions and associated institutional challenges for conservation banking in dynamic landscapes. *Landscape and Urban Planning*, 130, 64-72. <https://doi.org/10.1016/j.landurbplan.2014.06.004>
- Villarroya, A., Barros, A. C., & Kiesecker, J. (2014). Policy development for environmental licensing and biodiversity offsets in Latin America. *PLoS One*, 9(9), e107144. <https://doi.org/10.1371/journal.pone.0107144>
- Walz, U. (2015). Indicators to monitor the structural diversity of landscapes. *Ecological Modelling*, 295, 88-106. <https://doi.org/10.1016/j.ecolmodel.2014.07.011>
- Yu, S., Cui, B., & Gibbons, P. (2018). A method for identifying suitable biodiversity offset sites and its application to reclamation of coastal wetlands in China. *Biological Conservation*, 227, 284-291. <https://doi.org/10.1016/j.biocon.2018.09.030>
- zu Ermgassen, S. O. S. E., Baker, J., Griffiths, R. A., Strange, N., Struebig, M. J., & Bull, J. W. (2019). The ecological outcomes of biodiversity offsets under “no net loss” policies: A global review. *Conservation Letters*. <https://doi.org/10.1111/conl.12664>

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