Biological Conservation 155 (2012) 141–148

Contents lists available at SciVerse ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Faustian bargains? Restoration realities in the context of biodiversity offset policies

Martine Maron [a,*], Richard J. Hobbs [b], Atte Moilanen [c], Jeffrey W. Matthews [d,e], Kimberly Christie [b], Toby A. Gardner [f], David A. Keith [g], David B. Lindenmayer [h], Clive A. McAlpine [a,i]

[a] The University of Queensland, Landscape Ecology and Conservation Group, School of Geography, Planning and Environmental Management, Brisbane, Qld 4072, Australia
[b] School of Plant Biology, The University of Western Australia, 35 Stirling Highway, Crawley, WA 6009, Australia
[c] Finnish Centre of Excellence in Metapopulation Biology, Department of Biosciences, University of Helsinki, FIN-00014 Helsinki, Finland
[d] Illinois Natural History Survey, University of Illinois, Champaign, IL 61820, USA
[e] Department of Natural Resources and Environmental Sciences, University of Illinois, Urbana, IL 61801, USA
[f] Conservation Science Group, Department of Zoology, University of Cambridge, Downing Street, CB2 3EJ Cambridge, UK
[g] New South Wales Department of Environment and Climate Change, PO Box 1967, Hurstville, NSW 2220, Australia
[h] Fenner School of Environment and Society, ARC Centre of Excellence for Environmental Decisions and National Environmental Research Program, The Australian National University, Canberra, ACT 0200, Australia
[i] The University of Queensland, Environmental Decisions Group, Brisbane, Qld 4072, Australia

ARTICLE INFO

Article history:
Received 27 January 2012
Received in revised form 25 May 2012
Accepted 2 June 2012

Keywords:
Compensatory habitat
Conservation policy
Mitigation banking
Environmental risk
No net loss
Restoration success

ABSTRACT

The science and practice of ecological restoration are increasingly being called upon to compensate for the loss of biodiversity values caused by development projects. Biodiversity offsetting—compensating for losses of biodiversity at an impact site by generating ecologically equivalent gains elsewhere—therefore places substantial faith in the ability of restoration to recover lost biodiversity. Furthermore, the increase in offset-led restoration multiplies the consequences of failure to restore, since the promise of effective restoration may increase the chance that damage to biodiversity is permitted. But what evidence exists that restoration science and practice can reliably, or even feasibly, achieve the goal of 'no net loss' of biodiversity, and under what circumstances are successes and failures more likely? Using recent reviews of the restoration ecology literature, we examine the effectiveness of restoration as an approach for offsetting biodiversity loss, and conclude that many of the expectations set by current offset policy for ecological restoration remain unsupported by evidence. We introduce a conceptual model that illustrates three factors that limit the technical success of offsets: time lags, uncertainty and measurability of the value being offset. These factors can be managed to some extent through sound offset policy design that incorporates active adaptive management, time discounting, explicit accounting for uncertainty, and biodiversity banking. Nevertheless, the domain within which restoration can deliver 'no net loss' offsets remains small. A narrowing of the gap between the expectations set by offset policies and the practice of offsetting is urgently required and we urge the development of stronger links between restoration ecologists and those who make policies that are reliant upon restoration science.

1. Introduction

As the world's population passes seven billion, escalating conflicts between development and environmental conservation continue to diminish the Earth's stocks of natural capital. Projections suggest another 200 million to 1 billion hectares of terrestrial remnant vegetation will be converted for human land uses by 2050 (Millennium Ecosystem Assessment, 2005; Tilman et al., 2011). Biodiversity offsets (sometimes termed compensatory mitigation) are increasingly being used in an attempt to reduce this fundamental conflict between development (e.g. for mining, agriculture and urban development) and conservation (ten Kate et al., 2004; Kiesecker et al., 2009; McKenney and Kiesecker, 2010; Suding, 2011).

For the purposes of this paper, we define 'biodiversity offsetting' as compensating for losses of biodiversity components at an impact site by generating (or attempting to generate) ecologically equivalent gains, or 'credits', elsewhere (i.e. an offset site) (see Table 1 for definitions). As such, we consider only 'direct' offsets, rather than approaches to compensating for losses using indirect means, such as financial contributions not directly tied to generating ecologically equivalent biodiversity credits. Although some actions commonly referred to as 'biodiversity offsets' may not require demonstration of ecological equivalence of losses and gains, such equivalence is increasingly considered a fundamental aspect of the definition of a biodiversity offset (Business and Biodiversity Offsets Program, 2012).

* Corresponding author. Tel.: +61 7 3365 3836; fax: +61 7 3365 6899.
E-mail address: m.maron@uq.edu.au (M. Maron).

0006-3207/$ - see front matter © 2012 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.biocon.2012.06.003

Table 1
Definition of terms as used in this review.

Term	Definition
Biodiversity offsetting	The process of compensating for losses of biodiversity values at an impact site by generating ecologically equivalent gains, or 'credits', elsewhere (i.e. an offset site)
Biodiversity value	The aspect of biodiversity affected by the development or activity at the impact site, or generated at the offset site (e.g. a threatened species, a set of ecological functions, or a particular ecosystem type); often captured in a metric which combines information about condition and status
Biodiversity credit	A unit of a specified biodiversity value generated at an offset site to compensate for units of biodiversity lost at an impact site
Ecological equivalence	When the types of biodiversity values lost and gained are the same in nature and magnitude
Impact site	The site at which biodiversity values are lost or damaged
Offset site	The site at which additional biodiversity credits are generated through protection and/or restoration
Restoration	Activities aimed at increasing biodiversity values at a site, such as pest or weed control, management of regrowth vegetation, replanting of particular species, or implementation of a particular fire regime

Biodiversity offsets can be achieved in two main ways: (1) via averted loss from ongoing or anticipated impacts (e.g. avoided deforestation or degradation) at a site through the removal of threatening processes and (2) by enhancement of a degraded site through restoration and rehabilitation ('restoration offsets'). Averted loss can only generate 'gains' compared to a baseline of ongoing decline; restoration offsets are necessary if a cessation or reversal of biodiversity decline is to be achieved. In this review, we focus on restoration offsets and their potential to achieve genuine compensation for biodiversity losses.

A large range of restoration approaches is invoked in the context of offsets, including species, community and ecosystem-level interventions that vary from translocations of single taxa to multi-species introductions, ecosystem repair and generation of new ecosystems through revegetation (e.g. Harper and Quigley, 2005; Department of Sustainability and Environment, 2006; Gibbons and Lindenmayer, 2007). Biodiversity offsetting thus often relies heavily on restoration actions to generate biodiversity credits (to offset specific biodiversity losses or to trade for future losses, depending on the particular offset framework). Therefore, in many parts of the world, offset policies have become a significant driver of ecological restoration work (ten Kate et al., 2004; Robertson and Hayden, 2008; Palmer and Filoso, 2009).

Biodiversity offsetting may be conducted within a voluntary framework, with requirements negotiated between stakeholders, or within a statutory framework that is mandated by regional or national environmental legislation. Objectives vary among projects, but an increasingly cited goal is to achieve 'no net loss' or 'net gain' of biodiversity. Indeed, to avoid ambiguity and try and limit abuse of the term, the Business and Biodiversity Offsets Program (BBOP – http://.bbop.forest-trends.org/) considers no net loss as central to the definition of a biodiversity offset. The currency used to measure biodiversity losses and gains also varies, but may include particular ecological functions, size or viability of threatened species populations, and the extent and/or 'quality' of vegetation associations and habitat types. Commonly, an index based on a set of biodiversity attributes is used (e.g. the Habitat Hectares approach of Parkes et al., 2003). Usually, but not always, there is a requirement or preference for ecological equivalence—i.e., that gains must comprise the same type of biodiversity attributes that are lost (also called 'in kind' or 'like-for-like' offsets).

Such ambitious policy objectives as no net loss or net gain are often underpinned by the implicit belief that restoration ecologists and practitioners are, in general, able to restore or recreate ecosystems that contain equivalent biodiversity values to those that are lost. Yet restoration ecology is a relatively young and inexperienced discipline with a still-embryonic and patchy evidence base. Furthermore, given the complexity and variability of natural systems, the ecological community is increasingly recognizing that recreating or restoring ecosystems to some specified former state is often unlikely to be feasible (Hobbs et al., 2011), especially within reasonable time-frames. Thus, many current biodiversity offset approaches and expectations potentially push the limits of both scientific knowledge and practical feasibility (Stokstad, 2008; Palmer and Filoso, 2009; Hobbs et al., 2011).

In this paper we ask: to what extent are the demands that biodiversity offset policies make of restoration ecology realistic and feasible, given the state of current science? First, we briefly review recent growth in biodiversity offset-led restoration and its implications for restoration practice. Second, we examine the effectiveness of established biodiversity offset programs and review the current limits of restoration science. We then introduce a simple classification of the main sources of risk of failure in offsets from a restoration science perspective, and identify the types of biodiversity values for which offsetting may be: (a) feasible and low-risk, (b) higher risk and requiring of careful management, and (c) essentially unfeasible and inappropriate. Finally, we discuss potential responses to each of the risk factors, thereby helping to identify the domain in which restoration offsets may be effective mitigation tools.

2. The rapid expansion of offset-led restoration

The number and influence of biodiversity offset programs are growing rapidly. Madsen et al. (2010) identified 39 active biodiversity offset programs (i.e., comprising frameworks governing suites of individual offset projects) worldwide and 25 in some stage of development. The geographic reach of such programs is extensive. The regions that have most actively embraced biodiversity offsetting to date are North America and Australasia (with a combined total of 36 programs active or in development), although biodiversity offsetting is increasing in popularity elsewhere (Madsen et al., 2010). There are four active offset programs in Asia, (and another four in early development) resulting in the protection or restoration of approximately 26,000 hectares annually (Madsen et al., 2010). Many countries in South America have biodiversity offset-type programs at different stages of development, including the National Biodiversity Policy in Brazil, and '*Decreto 1753*' in Colombia, both of which include legislation outlining environmental mitigation principles (Madsen et al., 2010). South Africa has three offset policies being formulated, and although Europe has few programs in place, several are currently being piloted (including in the United Kingdom; DEFRA, 2011; Madsen et al., 2010). In addition to these government-mandated approaches, many companies undertake voluntary mitigation, particularly when operating in countries with limited legal protection for biodiversity (e.g. Tinto, 2004; Darbi et al., 2009; Newmont Golden Ridge Limited, 2009).

The proliferation of biodiversity offset programs and projects is driving a rapidly-growing demand for ecological restoration and management of newly-protected areas. Biodiversity offsetting under existing programs (encompassing a variety of definitions) is currently estimated to result in the protection or restoration of at least 86,000 hectares of land per year (Madsen et al., 2010). Between 1992/93 and 2001/02, the extent of wetlands restored or created in the US grew from 7148 hectares to 56,613 hectares (ten Kate et al., 2004). Combined, biodiversity offsets and wetland mitigation programs in the US alone have resulted in over 283,000 hectares of land protected or restored to 2008 (Madsen et al., 2010). Biodiversity offset activity is likely to continue to increase, in line with ongoing global development and economic growth (International Finance Corporation, 2006; Kiesecker et al., 2009). This growth in demand for biodiversity offsets is likely to be accompanied by an increase in financial resources available for restoration work.

3. Ecological effectiveness of biodiversity offsets

Wetland mitigation in the United States, which emerged in the 1970s and 1980s in response to Section 404 of the Clean Water Act (Hough and Robertson, 2009), is the policy for which most monitoring and evaluation data exist. Although not generally termed a 'biodiversity offset' approach, wetland mitigation nevertheless fits our broad definition (Table 1), as it aims to achieve no net loss of wetland values (including elements of biodiversity) and functions by generating wetland 'credits' through creation and restoration of wetland ecosystems (Corps and EPA, 2008).

Evaluating the effectiveness of wetland offsets is not straightforward. Offset sites are required to meet a set of performance criteria, usually established on a case-by-case basis and often based on local vegetation characteristics. However, these vegetation-based criteria have been criticized as vague and inadequate for ensuring that offset sites provide a genuine replacement for ecosystem functions lost when natural wetlands are destroyed (NRC, 2001). In response, federal regulatory agencies recently established a mitigation rule specifically requiring wetland mitigation projects to compensate for lost ecological functions (Corps and EPA, 2008), but it is not clear how this will be achieved in practice (Ruhl et al., 2009). An assortment of methods for rapid assessment of wetland functions has been developed and tested (Fennessy et al., 2007), but in the absence of an accepted method, losses and gains are primarily accounted for in terms of area of wetland and associated vegetation (Robertson, 2004).

Evidence from restored wetlands suggests that some ecosystem functions may take at least several decades to recover to a pre-disturbance state (Zedler and Callaway, 1999; BenDor, 2009). Some ecological indicators, including plant biomass and species richness, often recover rapidly in restored wetlands, but other important indicators, including species composition, soil physical and chemical properties, and ecosystem functions such as nutrient cycling, take much longer to be restored (e.g. Craft et al., 2003; Ballantine and Schneider, 2009; Gutrich et al., 2009; Hossler and Bouchard, 2010). For example, Hossler et al., (2011) found that, despite having similar vegetation and hydrology, restored and created wetlands stored significantly less C in soil and litter and had lower rates of denitrification than natural wetlands. In general, it cannot be assumed that restoration efforts will successfully return a degraded area to a state which is comparable or equivalent to the reference condition (Matthews and Spyreas, 2010). Mitigation wetlands are typically monitored on-site for three to five years after establishment (NRC, 2001). Therefore, many of the problems associated with wetland mitigation go undetected because they occur beyond the temporal scale of monitoring.

These challenges to successful wetland mitigation are similar to those faced by other types of environmental offsets. For example, Quigley and Harper (2006) found that at least 63% of projects designed to offset fish habitat loss in Canada failed to achieve the stated target of no net loss. This was because even when projects were fully compliant with legal practice standards, the restored systems remained functionally impoverished. Bernhardt and Palmer (2011) reviewed offset measures needed to compensate for the loss of over 1 million hectares of forest and 2000 km of streambed following extensive mining operations in the Appalachian Mountains, USA. They suggest that although the required stream reconstruction works may generate stable channels, there was no evidence that any of the approaches considered could replicate the ecological functions, such as maintenance of water quality, provided by the natural streams.

The lack of positive evaluations of ecological outcomes from biodiversity offset programs suggests that the approach deserves considerable further scrutiny. Are best practice techniques for restoration not being appropriately followed? Or are biodiversity offsets being used in situations where we simply lack the ability to restore the values in question?

4. Evidence from restoration science to date: what can we actually achieve?

Restoration activities have become a major part of ongoing efforts to better manage ecosystems and repair damage caused by past mismanagement and degradation (Hobbs and Harris, 2001; Hobbs and Cramer, 2008). However, there is ongoing debate about whether restoration can deliver successful outcomes given the current state of the science and practice (Hobbs et al., 2011). Part of this debate relates to how success is defined and hence the types of goals and outcomes expected from restoration projects (Hobbs, 2007). Success can be defined in many ways (Ruiz-Jaen and Aide, 2005), and because success or failure are hardly ever black and white concepts, restoration projects may succeed in achieving some goals but not others (Zedler, 2007). In addition, it is relatively difficult to obtain a clear picture of the frequency of success versus failure in projects from the growing body of literature on restoration because of limited monitoring and reporting (Bernhardt et al., 2005), under-reporting of failed projects (Hobbs, 2009) and the lack of robust evaluation frameworks for measuring success against ecological criteria (Gardner, 2010; Lindenmayer and Likens, 2010).

Suding (2011) recently reviewed successes and failures in restoration work in a variety of ecosystems worldwide and found that the level of success is highly dependent on geographic and historical context. Where restoration was being used to help the recovery of a degraded system, between a third and a half of projects reviewed were successful. However, where restoration aimed to generate new habitat, as is often the case with biodiversity offsets, success rates were lower still. Suding (2011) concluded that "…*although restoration is often possible and results in net positive benefits, it often does not go as well as planned. The inability to meet set criteria in many projects occurs at a high enough frequency to bring into question our ability to set realistic goals and our confidence in meeting these goals*".

In a survey of 87 restoration projects across a variety of terrestrial and aquatic ecosystem types and geographic locations globally, Lockwood and Pimm (1999) concluded that 17 were unsuccessful, 53 were partially successful and only 17 (20%) could be considered completely successful. They also examined the types of goals set for the restoration projects and categorized these goals as relating to functional attributes or structural and compositional attributes. Across all projects, goals relating to functional attributes were met in 61% of projects, partial return of structure/composition occurred in 66%, and full return of structure and species composition occurred in only 6%.

A more recent study by Rey Benayas et al. (2009) examined 89 published assessments of restoration projects, including examples from all inhabited continents. They considered each project's relative success in improving either biodiversity or ecosystem services both in comparison to the previous degraded state and in comparison to a reference system (representing the desired end point of the restoration). On average, restoration projects led to an improvement over the degraded situation in both biodiversity and ecosystem services, but did not approach the reference level for either.

Achieving different types of goals can often be a question of timescale. As such, it may be feasible to achieve some goals related to the recovery of some specific ecological functions more quickly than for goals related to the recovery of species composition. For instance, in a study of floodplain meadows in Europe, Woodcock et al. (2011) found that colonization by the majority of species that characterize the target habitat may take over 150 years, whereas functional trait structure can re-establish in less than half that time. They concluded that the time-scale needed to recreate grasslands calls into question the benefits of biodiversity offset approaches that allow grasslands to be lost to development under the presumption that their values can be recreated by restoration at other sites.

When habitat is re-created on a highly degraded site through revegetation, the revegetated site rarely resembles the target ecosystem. For example, Buckney and Morrison (1995) evaluated the success of revegetation treatments on mined Australian coastal sand plains. They showed that revegetated areas were on a trajectory toward development of a new ecological community that differed significantly in species composition from pre-mining vegetation and adjacent un-mined vegetation. Wilkins et al. (2003) and Lomov et al. (2009) analysed restoration trajectories of plants and invertebrates in restoration plantings on abandoned agricultural land, and came to the same conclusion. Similarly, Lindenmayer et al. (2012) found that replanted vegetation in agricultural areas of southern New South Wales supported a fundamentally different bird assemblage compared to old growth temperate woodlands and natural regrowth woodland. Similar results were found from work on reptiles in the same temperate woodland system (Cunningham et al., 2007; Michael et al., 2011). Indeed, after 10 years of detailed empirical work, it remained far from clear whether the recovery trajectory of restored areas would eventually lead to a reasonable level of congruence between the faunal assemblages of revegetation and original vegetation.

Achieving restoration success is also particularly challenging in situations that continue to be subject to external degrading influences, such as where permanent landscape changes such as urbanisation or agricultural intensification have occurred. For example, Stranko et al. (2012) examined the effectiveness of stream restoration in urban areas, and concluded that restoration activities were unable to improve any of the eight biodiversity indices they examined (Stranko et al., 2012). They concluded that the impacts of urbanisation on stream ecology were probably irreversible, and so the potential for biodiversity gains through restoration of degraded urban streams was limited.

From these studies and the restoration ecology literature in general, it is clear that some types of restoration are more likely to be successful than others. Recovery rates of different ecosystem types vary greatly, with or without restoration interventions (Holl and Aide, 2011; Jones and Schmitz, 2009). The type, extent, frequency and intensity of disturbance to which the system is exposed are important determinants of both the degree of intervention necessary and the likelihood of success. Hence, for instance, a cleared site where the biotic components are completely removed and the abiotic environment is significantly altered requires much greater levels of restoration input than an area that has been overgrazed, but is otherwise intact. Similarly, where particular biotic elements are missing, it may be relatively easy to reinstate these by controlling the factors leading to their demise. Relatively good success rates are possible for activities such as: (1) predator control, particularly in defined areas/islands (e.g. Moorhouse et al., 2003); (2) provision of specific resources for individual species (e.g. Gonzalez et al., 2006; Woodward et al., 2008); and (3) restoring native plant diversity and/or structural complexity by grazing removal in some systems (e.g. Pettit and Froend, 2001).

In contrast, success is less frequent for activities such as: (1) recreating a particular plant community or ecosystem type from a highly degraded state (Wilkins et al., 2003); (2) replacing 'full' floristic diversity (Munro et al., 2009) or restoring grassland in nutrient-enriched sites (Prober et al., 2002); and (3) restoring 'late successional' assemblages or 'old-growth' type habitat or habitat elements (Vesk et al., 2008; Lindenmayer and Wood, 2010; Maron et al., 2010). Legacies of past disturbance, multiple post-disturbance pathways, climate variability, and spatial and temporal variability all make achieving predictable restoration outcomes difficult (Mori, 2011), particularly where the restoration target is a complex biotic assemblage.

5. Limits to biodiversity offset effectiveness

The criticisms levelled at biodiversity offsetting are numerous, and relate to offset design, accounting, governance and compliance (Harper and Quigley, 2005; Gibbons and Lindenmayer, 2007; Walker et al., 2009; Bekessy et al., 2010). Here, we focus on the technical realities of restoration ecology as they affect the probability of offset success. We propose that the main factors limiting the ability of ecological restoration to achieve a successful offset are captured by the broad categories of poor measurability, uncertainty and time lags (Fig. 1).

5.1. Poor measurability

A fundamental problem in offsetting is the often poor definition and measurability of the value(s) to be offset (Walker et al., 2009; Bekessy et al., 2010; Quétier and Lavorel, 2011). Without being

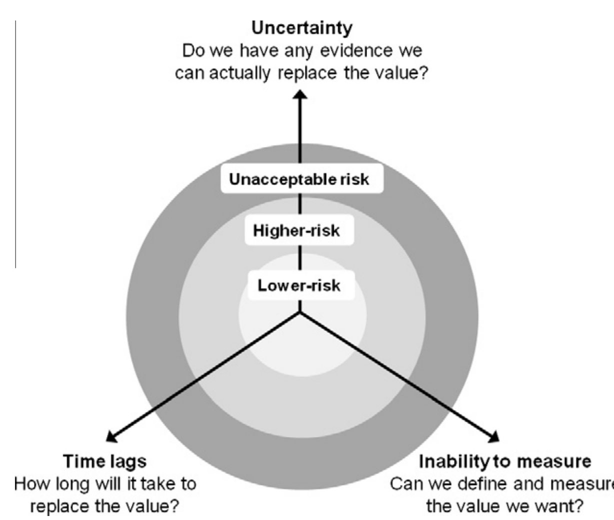

Fig. 1. Conceptual diagram representing three main factors (axes) that limit the technical effectiveness of biodiversity offsets. Axes represent: increasing uncertainty over our ability to restore; increasingly long expected time lags; and decreasing ability to define and measure the biodiversity value to be offset. As a proposed biodiversity offset moves along any one of these axes from the centre, it shifts from a domain within which there can be reasonable confidence in its success, through a domain in which offsetting entails a higher risk of failure and should trigger risk management responses, and finally to the range of values for which a successful offset outcome is highly unlikely, thus rendering offsetting inappropriate as a response to potential loss of that value. A given offset proposal may rank differently on each of the three axes.

able to define precisely (and then quantify) the values used in biodiversity loss–gain calculations, restoration efforts cannot be targeted and evaluated effectively (Bernhardt et al., 2005). Even when a precise identification of the value/s to be offset is available, it may not be possible to measure or monitor them accurately. These challenges of appropriate definition stem both from the inherent variability and complexity of the ecosystems being traded, but also from a lack of clarity around what biodiversity components we value the most.

In some cases, the biodiversity values can be defined precisely, quantified well, and often measured (or at least estimated) accurately (Fig. 1): for example, the number of individuals of a threatened species at a site. However, as the goal becomes more sophisticated (e.g., population viability of a threatened species) or aims to encompass more elements of an ecosystem (e.g., a plant community, or a set of ecological functions), measurability becomes more problematic. Increasingly, simplified metrics encapsulating multiple values are being used as offset currency (e.g. Parkes et al., 2003; Gibbons et al., 2009), but these necessarily increase the risk of offsets failing to meet the 'like for like' criteria because losses or gains in individual components can be masked within the single value of the metric, or because the metric itself does not include important values, such as ecosystem function (Palmer, 2009).

5.2. Uncertainty in restoration outcomes

One of the most common criticisms levelled at biodiversity offsets is that they exchange certain losses for uncertain gains. Understanding the effectiveness of a restoration project, and the timescale over which expected benefits will be accrued, is fraught with uncertainty.

Uncertainty of outcomes is particularly high when an offset depends upon the restoration of significantly modified sites (Hilderbrand et al., 2005). Relative uncertainty may be lower where the offset involves the removal of a threatening or degrading process, such as the control of an invasive species (Hilderbrand et al., 2005). For example, local populations of *Callaeas cinerea wilsoni*, a New Zealand bird, recovered within three years of the commencement of management to control mammalian pests (Innes et al., 1999). The identification of predation as a factor limiting a species' population size may therefore allow reasonably high confidence in offsets involving pest management. However, if the goal is to restore a degraded woodland plant community to something structurally and compositionally similar to a reference site, then success is far less certain (Wilkins et al., 2003). Uncertainty in outcomes can be further exacerbated by the potential for interaction effects from background climate variability and environmental change (Harris et al., 2006). The less certain we are that we possess the knowledge and technical ability to restore a biodiversity value, the less appropriate is offsetting as a response to potential loss of the value (Fig. 1).

5.3. Time lags

Even if an offset goal is measurable and the uncertainty of it being achieved is low, there are often unavoidable time lags before the goal is realized (Zedler, 1996; Hilderbrand et al., 2005; Morris et al., 2006). Offsets should account for these time-lags, because it is not considered fair to compensate immediate loss by hypothetical equal gains in the distant future (Norton, 2009; Moilanen et al., 2009; Quétier and Lavorel, 2011). In some cases, a restoration action may have an almost immediate effect—for example, a species may be known to use artificial nest hollows as readily as natural ones, and so the hollows can be provided as soon as the natural ones are lost. By contrast, replanting seedling feed trees for a species to compensate for the loss of mature feed trees has a relatively high probability of success but may only be achieved after many decades (Maron et al., 2010). In such circumstances, interim supplementation of the affected resource may be an important component of an offset, although the temporal deficit may be impossible to compensate (Moilanen et al., 2009). Long time-lags may also result in severe resource bottlenecks, during which a target species or community suffers increased vulnerability to other threats. When time-lags are unacceptably long, even high confidence in the ability to restore a value eventually does not reduce risk to an acceptable level (Fig. 1).

6. Improving risk management in biodiversity offsetting

Given the challenges to effective use of biodiversity offsets, the domain within which offsetting is an appropriate response to threats to biodiversity values is limited. Nevertheless, there are ways in which risks to offset success can be better managed. Below, we briefly explore how these approaches can help to manage the technical challenges of poor measurability, uncertainty and time lags.

6.1. Responses to poor measurability

Improving measurability requires the development of better habitat metrics, biodiversity indicators and surrogates. Objectives should be clear in terms of which biodiversity values an offset should target, and metrics and monitoring programs designed accordingly (Bekessy et al., 2010). Ideally, multiple aspects of the value to be offset should be measured and monitored to provide a more informative record of offset performance. However, despite increased research attention, fundamental problems remain, such as how to quantify the contribution of candidate offset sites to wider landscape connectivity or regional-scale ecological processes. The more difficult it is to define and/or measure the biodiversity value targeted, the less we can claim to know about the success of restoration attempts.

6.2. Responses to uncertainty

Multipliers are commonly proposed as a way of dealing with uncertainty in outcomes at an offset site (Dunford et al., 2004; Bruggeman et al., 2005). A multiplier should be scaled to the degree of uncertainty in the effectiveness of the offset activity. Yet this is rarely done in offset policies. Where multipliers are used, they are often intended to reflect conservation significance of the biodiversity values in question, and the justification for their value is often unclear. Moilanen et al. (2009) investigated what they call a "fair offset ratio"—the level at which a multiplier provides a robust guarantee of a favourable outcome. Simulation analyses revealed that a comprehensive accounting of uncertainty can result in very large multipliers, which in many cases would be politically and economically unacceptable. Moreover, in a practical setting, such a quantitative assessment of uncertainty is often impossible given the lack of information about the ecology of the biodiversity values in question and the effectiveness of potential interventions.

One response to this challenge of uncertainty is to invest efforts not in active restoration, but in averting further losses (through improved protection) of existing yet threatened areas that can then be used as (averted loss) offset credits. Although this strategy has the advantage of not relying on a highly uncertain ability to re-create biodiversity values, there are limited circumstances under which averted loss can be considered to represent true additionality (particularly in nations with well-developed biodiversity protection controls), and estimates of this additionality are themselves subject to significant uncertainty (Gibbons and Lindenmayer, 2007). This is because the approach relies on accurate estimation of the probability of loss of biodiversity values at the offset site in the absence of the additional protection (Maron et al., 2010). It therefore implies acceptance of a baseline of continuing biodiversity decline under current policy settings. Even if these requirements are met, the use of averted loss as an offset can introduce a conundrum. Offset policies frequently permit the 'protection' of a site as an averted loss offset, even if loss of the offset site itself would have had to be offset. This is one of several concerns over the current use of averted loss credits in offsetting.

A commitment to active adaptive management (McKenney and Kiesecker, 2010; Gardner, 2010) can help to resolve uncertainties in achieving restoration offset goals. This involves setting dual objectives for both restoration and learning at the outset of restoration offset projects. Key elements of this approach include experimental design to compare alternative strategies, monitoring to compare their relative merits and adjustment of strategies based on new knowledge that emerges from the restoration experiment (Keith et al., 2011). Without such comparative experimentation, opportunities for learning are limited and uncertainty about outcomes may not be reduced (Walters and Holling, 1990). Simultaneous exploration of multiple restoration options also spreads the risk of failure more widely than if all resources were channelled into a single option. Unfortunately, most restoration projects simply implement current best practice (a single management option), are often spatially unreplicated, and outcomes are monitored until failure or more fashionable options emerge. Although optimal experimental designs may not often be feasible in an offsetting context, imaginative synthesis across restoration projects can generate robust designs that reduce future uncertainties about restoration success (Keith et al., 2011).

Because offset policies raise the stakes involved in restoration projects, there is a clear need for greater investment in restoration ecology research. Already several offset policies specify options for contributing financially to relevant research. In cases where knowledge is too limited to implement an offset with confidence, it may be argued that the financial burden of generating the required knowledge should fall to the proponents of the development project that triggers the offset. However, it cannot be argued that this contribution, in itself, constitutes an offset—it is merely a necessary step enabling an offset. Restoration ecologists, too, must engage more constructively and effectively with policy makers to ensure that the questions being tackled are those most likely to be useful to the biodiversity markets of the future.

6.3. Responses to time lags

Several authors have promoted the idea that 'biodiversity banks' of already-accrued credits (whether through restoration or averted risk) should be generated before biodiversity values are lost (e.g. Bekessy et al., 2010). In theory, such an approach could eliminate the problems associated with long time lags in restoration and uncertainty of offset outcomes. One criticism of this approach, however, relates to the problem of accurately measuring additionality. In countries such as Australia, for example, much restoration activity or land management above 'duty of care' is already done on a voluntary basis by individual landholders and community groups. If changes to offset policies mean that voluntary restoration activities are now considered to have generated saleable biodiversity credits, this is likely to present a difficult-to-resist temptation: to take the opportunity to sell the credits generated, despite the fact that such credits can then be used to trade for biodiversity destruction elsewhere. Thus, if restoration actions that would have been done outside of biodiversity markets are now used generate biodiversity credits for offsetting, genuine additionality will be eroded.

An alternate solution to banking of credits, likely to be more workable in situations which necessarily involve time lags and uncertainty, is to require the proponents of the development activity to purchase insurance that covers the risk of offset failure. Any such approach relies on the biodiversity values to be traded being clearly defined and measurable, and raises the problem of how premiums might be used to deliver required outcomes in the instance of failure. If restoration happens with a time delay and failure thus also only becomes apparent after a time delay, there will be counterparty risk about the ability of the insurance provider to make good on the insurance. Nevertheless, the development of an insurance market for biodiversity would increase pressure for clarity around policy requirements and would introduce additional incentives to avoid high-risk trades.

Finally, time discounting is an easily (but often poorly) implemented method originating from economics that can be used to value future gains in present-time units, as well as account for risk (Carpenter et al., 2007). For example, habitat equivalency analysis, an approach used to quantify ecological losses and gains, includes time discounting as an option (Dunford et al., 2004; Bruggeman et al., 2005). Implementation of time discounting requires robust estimates of the ecological time lags. These may be obtained either by observation and projection from existing time series or by mechanistic modelling based on an understanding of the processes involved. The influence of time discounting on the fair offset ratio (the ratio of the quantum of offset activity to the quantum of initial impact that results in a fair trade of biodiversity) may be very large (Moilanen et al., 2009). In fact, if the development of biodiversity value is very slow, it is questionable whether the value should be considered restorable at all in an ecological sense (Morris et al., 2006).

7. Conclusion

Confidence in the ability of restoration to deliver genuine biodiversity offsets is undermined by the problems of defining and measuring the biodiversity values that are lost and gained, considerable uncertainty surrounding the effectiveness of restoration techniques, and long time-lags. The increasingly broad application of offsetting, often with limited scientific support, is therefore of concern (Palmer and Filoso, 2009). We recommend that restoration be used to deliver biodiversity offsets only when: (1) the impacted biodiversity and ecosystem values can be explicitly defined and measured; (2) there is an existing and sound evidence base that restoration of the values in question is feasible; and (3) time lags and uncertainties involved are explicitly accounted for in a loss–gain calculation, and any time lags do not pose an interim threat to the persistence of the biodiversity value in question. A plea for policy makers to operate within the domain of scientifically realistic options is hardly new. Nevertheless, the rapidly-increasing reach of biodiversity offsetting into many areas of environmental policy—including threatened species protection, environmental impact assessment and protected area investment—makes closer collaboration between policy makers and restoration scientists and practitioners an urgent priority (Palmer, 2009).

Acknowledgements

This contribution originated from a workshop held by the ARC Centre for Excellence in Environmental Decisions. The South African National Biodiversity Institute and the University of Cambridge kindly hosted MM during preparation of part of the manuscript. RJH acknowledges support from an ARC Australian Laureate Fellowship. AM was supported by the ERC-StG project GEDA, Grant 260393. CAM is supported by an ARE Future Fellowship. TAG thanks the Natural Environment Research Council, UK for funding (NE/FO1614X/1).

References

Ballantine, K., Schneider, R., 2009. 55 Years of soil development in restored freshwater depressional wetlands. Ecol. Appl. 19, 1467–1480.

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. Conserv. Lett. 3, 151–158.

BenDor, T., 2009. A dynamic analysis of the wetland mitigation process and its effects on no net loss policy. Landscape Urban Plan. 89, 17–27.

Bernhardt, E.S., Palmer, M.A., 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. In: Ostfeld, R.S., Schlesinger, W.H. (Eds.), Year in Ecology and Conservation Biology, pp. 39–57.

Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B., Sudduth, E., 2005. Synthesizing US river restoration efforts. Science 308, 636–637.

Bruggeman, D.J., Jones, M.L., Lupi, F., Scribner, K.T., 2005. Landscape equivalency analysis: methodology for estimating spatially explicit biodiversity credits. Environ. Manage. 36, 518–534.

Buckney, R.T., Morrison, D.A., 1995. Temporal trends in plant species composition on mined sand dunes in Myall Lakes National Park, Australia. Aust. J. Ecol. 17, 241–254.

Business and Biodiversity Offsets Programme (BBOP), 2012. Standard on Biodiversity Offsets. BBOP, Washington, DC. <http://www.bbop.forest-trends.org/guidelines/Standard.pdf>.

Carpenter, S.R., Brock, W.A., Ludwig, D., 2007. Appropriate discounting leads to forward-looking ecosystem management. Ecol. Res. 22, 10–11.

Corps (US Army Corps of Engineers), EPA (US Environmental Protection Agency), 2008. Compensatory mitigation for losses of aquatic resources. Federal Register, pp. 19594–19705.

Craft, C., Megonigal, P., Broome, S., Stevenson, J., Freese, R., Cornell, J., Zheng, L., Sacco, J., 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. Ecol. Appl. 13, 1417–1432.

Cunningham, R.B., Lindenmayer, D.B., Crane, M., Michael, D., MacGregor, C., 2007. Reptile and arboreal marsupial response to replanted vegetation in agricultural landscapes. Ecol. Appl. 17, 609–619.

Darbi, M., Ohlenburg, H., Herberg, A., Wende, W., Skambracks, D., Herbert, M., 2009. International approaches to compensation for impacts on biological diversity. Final Report, Berlin.

Defra., 2011. Biodiversity Offsetting. Technical Paper: proposed metric for the biodiversity offsetting pilot in England. Department of Environment, Food and Rural Affairs, London.

Department of Sustainability and Environment, 2006. Native Vegetation Gain Approach – Technical Basis for Calculating Gains Through Improved Native Vegetation Management and Revegetation. Victorian Government, Department of Sustainability and Environment, East Melbourne.

Dunford, R.W., Ginn, T.C., Desvousges, W.H., 2004. The use of habitat equivalency analysis in natural resource damage assessments. Ecol. Econ. 48, 49–70.

Fennessy, M.S., Jacobs, A.D., Kentula, M.E., 2007. An evaluation of rapid methods for assessing the ecological condition of wetlands. Wetlands 27, 543–560.

Gardner, T.A., 2010. Monitoring Forest Biodiversity: Improving Conservation through Ecologically-Responsible Management. Earthscan, London.

Gibbons, P., Lindenmayer, D.B., 2007. Offsets for land clearing: no net loss or the tail wagging the dog? Ecol. Manage. Restor. 8, 26–31.

Gibbons, P., Briggs, S.V., Ayers, D., Seddon, J., Doyle, S., Cosier, P., McElhinny, C., Pelly, V., Roberts, K., 2009. An operational method to assess impacts of land clearing on terrestrial biodiversity. Ecol. Ind. 9, 26–40.

Gonzalez, L., Margalida, A., Sanchez, R., Oria, J., 2006. Supplementary feeding as an effective tool for improving breeding success in the Spanish imperial eagle (*Aquila adalberti*). Biol. Conserv. 129, 477–486.

Gutrich, J.J., Taylor, K.J., Fennessy, M.S., 2009. Restoration of vegetation communities of created depressional marshes in Ohio and Colorado (USA): the importance of initial effort for mitigation success. Ecol. Eng. 35, 351–368.

Harper, D.J., Quigley, J.T., 2005. No net loss of fish habitat: a review and analysis of habitat compensation in Canada. Environ. Manage. 36, 343–355.

Harris, J.A., Hobbs, R.J., Higgs, E., Aronson, J., 2006. Ecological restoration and global climate change. Restor. Ecol. 14, 170–176.

Hilderbrand, R.H., Watts, A.C., Randle, A.M., 2005. The myths of restoration ecology. Ecology and Society 10, 19.

Hobbs, R.J., 2007. Setting effective and realistic restoration goals: key directions for research. Restor. Ecol. 15, 354–357.

Hobbs, R., 2009. Looking for the silver lining: making the most of failure. Restor. Ecol. 17, 1–3.

Hobbs, R.J., Cramer, V.A., 2008. Restoration ecology: interventionist approaches for restoring and maintaining ecosystem function in the face of rapid environmental change. In: Annual Review of Environment and Resources. pp. 39–61.

Hobbs, R.J., Harris, J.A., 2001. Restoration ecology: repairing the Earth's ecosystems in the new millennium. Restor. Ecol. 9, 239–246.

Hobbs, R.J., Hallett, L.M., Ehrlich, P.R., Mooney, H.A., 2011. Intervention ecology: applying ecological science in the twenty-first century. Bioscience 61, 442–450.

Holl, K.D., Aide, T.M., 2011. When and where to actively restore ecosystems? For. Ecol. Manage. 261, 1558–1563.

Hossler, K., Bouchard, V., 2010. Soil development and establishment of carbon-based properties in created freshwater marshes. Ecol. Appl. 20, 539–553.

Hossler, K., Bouchard, V., Fennessy, M.S., Frey, S.D., Anemaet, E., Herbert, E., 2011. No-net-loss not met for nutrient function in freshwater marshes: recommendations for wetland mitigation policies. Ecosphere 2, art82.

Hough, P., Robertson, M., 2009. Mitigation under Section 404 of the Clean Water Act: where it comes from, what it means. Wetlands Ecol. Manage. 17, 15–33.

Innes, J., Hay, R., Flux, I., Brad, P., Speed, H., Jansen, P., 1999. Successful recovery of North Island kokako *Callaeas cinerea wilsoni* populations, by adaptive management. Biol. Conserv. 87, 201–214.

International Finance Corporation, 2006. Performance Standard 6: Biodiversity Conservation and Sustainable Natural Resource Management. International Finance Corporation, World Bank Group. <http://www.ifc.org/ifcext/sustainability.nsf/Content/PerformanceStandards>.

Jones, H.P., Schmitz, O.J., 2009. Rapid recovery of damaged ecosystems. PLoSOne 4, e5653.

Keith, D.A., Martin, T.G., McDonald-Madden, E., Walters, C., 2011. Uncertainty and adaptive management for biodiversity conservation. Biol. Conserv. 144, 1175–1178.

Kiesecker, J.M., Copeland, H., Pocewicz, A., Nibbelink, N., McKenney, B., Dahlke, J., Holloran, M., Stroud, D., 2009. A framework for implementing biodiversity offsets: selecting sites and determining scale. Bioscience 59, 77–84.

Lindenmayer, D.B., Likens, G.E., 2010. Effective Ecological Monitoring. CSIRO Publishing, Melbourne and Earthscan, London.

Lindenmayer, D.B., Wood, J.T., 2010. Long-term patterns in the decay, collapse, and abundance of trees with hollows in the mountain ash (*Eucalyptus regnans*) forests of Victoria, southeastern Australia. Can. J. For. Res. 40, 48–54.

Lindenmayer, D.B., Northrop-Mackie, A.R., Montague-Drake, R., Michael, D., Crane, M., Okada, S., MacGregor, C., Gibbons, P., 2012. Not all kinds of regrowth are created equal: regrowth type influences bird assemblages in threatened Australian woodland ecosystems. PLoS ONE 7 (4), e34527.

Lockwood, J.L., Pimm, S.L., 1999. When does restoration succeed? In: Weiher, E., Keddy, P. (Eds.), Ecological Assembly Rules: Perspectives, Advances, Retreats. Cambridge University Press, Cambridge.

Lomov, B., Keith, D.A., Hochuli, D.F., 2009. Linking ecological function to species composition in ecological restoration: seed removal by ants in recreated woodland. Austral Ecol. 34, 751–760.

Madsen, B., Carroll, N., Moore Brands, K., 2010. State of biodiversity markets report: offset and compensation programs worldwide. <http://www.ecosystemmarketplace.com/documents/acrobat/sbdmr.pdf>.

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. J. Appl. Ecol. 47, 348–355.

Matthews, J.W., Spyreas, G., 2010. Convergence and divergence in plant community trajectories as a framework for monitoring wetland restoration progress. J. Appl. Ecol. 47, 1128–1136.

McKenney, B.A., Kiesecker, J.M., 2010. Policy development for biodiversity offsets: a review of offset frameworks. Environ. Manage. 45, 165–176.

Michael, D.R., Cunningham, R.B., Lindenmayer, D.B., 2011. Regrowth and revegetation in temperate Australia presents a conservation challenge for reptile fauna in agricultural landscapes. Biol. Conserv. 144, 407–415.

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Scenarios. Island Press, Washington, DC.

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. Restor. Ecol. 17, 470–478.

Moorhouse, R., Greene, T., Dilks, P., Powlesland, R., Moran, L., Taylor, G., Jones, A., Knegtmans, J., Wills, D., Pryde, M., Fraser, I., August, A., August, C., 2003. Control of introduced mammalian predators improves kaka Nestor meridionalis breeding success: reversing the decline of a threatened New Zealand parrot. Biol. Conserv. 110, 33–44.

Mori, A.S., 2011. Ecosystem management based on natural disturbances: hierarchical context and non-equilibrium paradigm. J. Appl. Ecol. 48, 280–292.

Morris, R.K.A., Alonso, I., Jefferson, R.G., Kirby, K.J., 2006. The creation of compensatory habitat – can it secure sustainable development? J. Nat. Conserv. (Jena) 14, 106–116.

Munro, N.T., Fischer, J., Wood, J., Lindenmayer, D.B., 2009. Revegetation in agricultural areas: the development of structural complexity and floristic diversity. Ecol. Appl. 19, 1197–1210.

Newmont Golden Ridge Limited, 2009. BBOP Pilot Project Case Study. Akyem Gold Mining Project, Eastern Region, Ghana. Accra, Ghana. <http://www.forest-trends.org/biodiversityoffsetprogram/guidelines/newmont-casestudy.pdf>.

Norton, D.A., 2009. Biodiversity offsets: two New Zealand case studies and an assessment framework. Environ. Manage. 43, 698–706.

NRC (National Research Council), 2001. Compensating for wetland losses under the Clean Water Act. National Academy Press, Washington DC.

Palmer, M.A., 2009. Reforming watershed restoration: science in need of application and applications in need of science. Estuaries Coasts 32, 1–17.

Palmer, M.A., Filoso, S., 2009. Restoration of ecosystem services for environmental markets. Science 325, 575–576.

Parkes, D., Newell, G., Cheal, D., 2003. Assessing the quality of native vegetation: the 'habitat hectares' approach. Ecol. Manage. Restor. 4, S29–S38.

Pettit, N.E., Froend, R.H., 2001. Long-term changes in the vegetation after the cessation of livestock grazing in *Eucalyptus marginata* (jarrah) woodland remnants. Austral Ecol. 26, 22–31.

Prober, S.M., Thiele, K.R., Lunt, I.D., 2002. Identifying ecological barriers to restoration in temperate grassy woodlands: soil changes associated with different degradation states. Aust. J. Bot. 50, 699–712.

Quétier, F., Lavorel, S., 2011. Assessing ecological equivalence in biodiversity offset schemes: key issues and solutions. Biol. Conserv. 144, 2991–2999.

Quigley, J.T., Harper, D.J., 2006. Effectiveness of fish habitat compensation in Canada in achieving no net loss. Environ. Manage. 37, 351–366.

Rey Benayas, J.M., Newton, A.C., Diaz, A., Bullock, J.M., 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science 325, 1121–1124.

Robertson, M.M., 2004. The neoliberalization of ecosystem services: wetland mitigation banking and problems in environmental governance. Geoforum 35, 361–373.

Robertson, M., Hayden, N., 2008. Evaluation of a market in wetland credits: entrepreneurial wetland banking in Chicago. Conserv. Biol. 22, 636–646.

Ruhl, J.B., Salzman, J., Goodman, I., 2009. Implementing the new ecosystem services mandate: a catalyst for advancing science and policy. Natl. Wetlands Newslett. 31 (11–13), 20.

Ruiz-Jaen, M.C., Aide, T.M., 2005. Restoration success: how is it being measured? Restor. Ecol. 13, 569–577.

Stokstad, E., 2008. Environmental regulation: new rules on saving wetlands push the limits of the science. Science 320, 162–163.

Stranko, S.A., Hilderbrand, R.H., Palmer, M.A., 2012. Comparing the fish and benthic macroinvertebrate diversity of restored urban streams to reference streams. Restor. Ecol.. http://dx.doi.org/10.1111/j.1526-100X.2011.00824.x.

Suding, K.N., 2011. Toward an era of restoration in ecology: successes, failures and opportunities ahead. Annu. Rev. Ecol. Evol. Syst. 42, 465–487.

ten Kate, K., Bishop, J., Bayon, R., 2004. Biodiversity Offsets: Views, Experience, and the Business Case. IUCN: Gland Switzerland and Cambridge UK, Insight Investment: London UK.

Tilman, D., Balzer, C., Hill, J., Befort, B.L., 2011. Global food demand and the sustainable intensification of agriculture. PNAS 108, 20260–20264.

Tinto, Rio, 2004. Rio Tinto's Biodiversity Strategy. UK and Melbourne, Australia, London.

Vesk, P.A., Nolan, R., Thomson, J.R., Dorrough, J.W., Mac Nally, R., 2008. Time lags in provision of habitat resources through revegetation. Biol. Conserv. 141, 174–186.

Walker, S., Brower, A.L., Stephens, R.T.T., Lee, W.G., 2009. Why bartering biodiversity fails. Conserv. Lett. 2, 149–157.

Walters, C.J., Holling, C.S., 1990. Large-scale management experiments and learning by doing. Ecology 71, 2060–2068.

Wilkins, S., Keith, D.A., Adam, P., 2003. Measuring success: evaluating the restoration of a grassy eucalypt woodland on the Cumberland Plain, Sydney, Australia. Restor. Ecol. 11, 489–503.

Woodcock, B.A., McDonald, A.W., Pywell, R.F., 2011. Can long-term floodplain meadow recreation replicate species composition and functional characteristics of target grasslands? J. Appl. Ecol. 48, 1070–1078.

Woodward, W., Ellis, W.A., Carrick, F.N., Tanizaki, M., Bowen, D., Smith, P., 2008. Koalas on North Stradbroke Island: diet, tree use and reconstructed landscapes. Wildlife Res. 35, 606–611.

Zedler, J.B., 1996. Ecological issues in wetland mitigation: an introduction to the forum. Ecol. Appl. 6, 33–37.

Zedler, J.B., 2007. Success: an unclear, subjective descriptor of restoration outcomes. Ecol. Restor. 25, 162–168.

Zedler, J.B., Callaway, J.C., 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories? Restor. Ecol. 7, 69–73.