A review of biodiversity offsets implemented in the Australian Capital Territory under the *Environment Protection and Biodiversity Conservation Act 1999*

by

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Candidate's Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

Brooke Connors Date: 30/05/2019

Acknowledgements

I'm ecstatic that I have been given the opportunity to undertake Honours at the Fenner School. After undergrad, I was entirely convinced that further study wasn't up my alley. A year working full-time was enough 'real-world' experience to change my mind…

It has been an ~ honour ~

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Abstract

Biodiversity offsetting is a widely adopted policy mechanism that attempts to balance the environmental impacts at one site, with promised gains at another. In theory, offsets are able to compensate for irreversible environmental impacts by protecting and/or enhancing biodiversity. However, studies indicate that there are several issues and risks associated with the application of offset policies. Since predicted future trends of population growth, urbanisation and agricultural production indicate that adverse environmental impacts are likely to intensify, we can expect that slowing biodiversity loss will remain a key global challenge. It is vital that the mechanisms intended to balance biodiversity impacts, such as biodiversity offsetting, are subject to robust evaluations. Otherwise, it is difficult to preform adequate policy learning and adaption.

In this thesis, I provide the first review of offsets delivered under Australia's federal biodiversity offset policy by exploring the extent to which three key offsetting principles are being achieved. To do this, I focus on offsets implemented in the Australian Capital Territory (ACT) and evaluate the extent to which these offsets are: (1) equivalent, or 'like-for-like', (2) have achieved 'no net loss', and (3) are additional. Using the limited data available, I demonstrate:

1. In attempt to make losses and gains equivalent, simplification and substitution may be occurring, in particular:
    a. I observed a greater amount of native grassland was lost at development sites than gained through offsetting, which could impact on grassland reliant fauna.
    b. At the landscape scale, my analysis suggests that development sites usually occur in more fragmented landscapes while offsets occur in more intact. Because of this difference, I predicted a greater impact per ha on species richness from development than gained through offsetting.
2. It is only possible to achieve no net loss within a suitable timeframe when offsets are largely based on restoration (rather than 'averted losses') and restoration success is infeasibly high. I also consider the implications of applying averted loss in the context of recent guidance.
3. There are two key issues hindering the successful implementation and evaluation of additionality:
    a. The EPBC offset policy frames additionality in a way that confuses the aim of this principle and incorrectly encourages the calculation of gains from averted loss.
    b. There is inadequate transparency regarding site counterfactual scenarios. Particularly, the existing financial and management commitments within various land-use zones and protected areas which offsets have been applied.

In light of these findings, I propose key areas for future research and highlight recommendations based on empirical evaluations and literature to date.

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EPBC: *Environment Protection and Biodiversity Conservation Act*

MNES: Matters of national environmental significance

Chapter 1: Introduction

Despite global efforts, the rate of biodiversity loss shows no signs of slowing (Butchart *et al.*, 2010; Díaz *et al.*, 2019). Between the years 2000 and 2013 alone, approximately 2.3 million km^2 of tree cover was lost worldwide (Hansen *et al.*, 2013b). Equally alarming is the acceleration of species extinction, which is estimated at 10's to 100's times higher than the background rate (Díaz *et al.*, 2019; Pimm *et al.*, 2014). Slowing biodiversity loss is essential to sustaining healthy ecosystem processes, and thus, humanity (Cardinale *et al.*, 2012; Chapin *et al.*, 2000; Maxwell *et al.*, 2016; Díaz *et al.*, 2019). However, it remains a complex challenge for three key reasons: (1) the main drivers of biodiversity loss (i.e. overexploitation, agriculture and urban development) are integral to sustaining livelihoods and economies (Maxwell *et al.*, 2016), (2) predicted future trends of population growth and urbanisation indicate that these drivers are likely to intensify (Maxwell *et al.*, 2016; United Nations, 2017; Food and Agriculture Organization of the United Nations, 2018; United Nations, 2018) and (3), so far, attempts made to slow (or halt) habitat loss and safeguard biodiversity have proved unsuccessful (Butchart *et al.*, 2010; Hoffmann *et al.*, 2010; Díaz *et al.*, 2019).

Since the 1970s, biodiversity conservation endeavours have expanded from traditional protection (i.e. national parks) and 'command and control' regulations to include a variety of policy instruments (Evans, 2017; Sullivan, 2005; Bonneuil, 2015). This expansion included the development of biodiversity offsetting (Bonneuil, 2015). Biodiversity offsets seek to balance the residual environmental impacts of projects (e.g. development) after appropriate measures to avoid and minimise impacts have been taken (Business and Biodiversity Offsets Programme, 2013). So, the fundamental assumption of biodiversity offsetting is that the losses at one site can be made equivalent to the potential gain at another. Even though the global uptake of offset policies has spiked in the last decade (Maron *et al.*, 2016b), there is a considerable lack of reporting on what outcomes are being delivered and this is recognised as a current, ongoing challenge (Gibbons *et al.*, 2018; Carver and Sullivan, 2017).

Evaluating policy interventions that involve biodiversity conservation is recognised as a complex endeavour (Ferraro and Pattanayak, 2006). Particularly complicated is evaluating biodiversity offsetting since there is no 'standard' approach to these policies (Miller *et al.*, 2015). As a policy mechanism, offsetting also has many prevailing issues that span ethical, social, technical, and governance domains (Maron *et al.*, 2016b). In spite of these challenges, empirical evaluations are emerging in the literature at both policy level (Gibbons *et al.*, 2018; May *et al.*, 2017; Bezombes *et al.*, 2019), and site level (Thorn *et al.*, 2018; Lindenmayer *et al.*, 2017). Yet, such evaluations are limited when compared to the expansive amount of theoretical literature and broad application of biodiversity offsetting policies (Maron *et al.*, 2016b). In order to support successful policy learning and adaption, it is vital that greater efforts are made to deliver empirical evaluations (Dovers and Hussey, 2013). Otherwise it is difficult to understand the present impacts of these policies and identify key areas for improvement.

While Australia has a moderately small population (reaching 25 million in mid-2018) (Australian Bureau of Statistics, 2019), it has a native vegetation clearing rate described as globally significant (Evans, 2016); boasting a loss of approximately 44% of the nation's forests and woodlands since European settlement (Jackson *et al.*, 2017). There is also a lack of evidence that Australia's native vegetation policies have been effective in reducing land clearing (Evans, 2016). As a result, Australia's unique biodiversity continues to be threatened by habitat loss and fragmentation associated with clearing (Australian State of the Environment Committee, 2001; Cresswell and Murphy, 2017). The majority of recent land clearing in Australia is for conversion to pasture (Evans, 2016), which is unsurprising since Australia's agricultural industry accounts for 58% of total land use (Australian Bureau of Agricultural and Resource Economics and Sciences, 2018b). While resulting in less vegetation clearing by magnitude, urban expansion causes irreversible and severe impacts in areas that are often already highly fragmented. Since habitat for more than 50% of Australia's threatened species occurs in and around major cities, the expansion of urban areas is considered a major threat to Australia's biodiversity (Yencken and Wilkinson, 2000).

Australia's key environmental legislation is the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), which came into effect in 2000. The nation's biodiversity offset policy was formally introduced under this Act in 2012, namely, *The Environment Protection and Biodiversity Conservation Act Environmental Offset Policy* (EPBC offset policy) (Miller *et al.*, 2015). While there has been a senate inquiry into the 'history, appropriateness and effectiveness of the use of environmental offsets' required under the EPBC Act (Parliament of Australia, 2014) there has not been any empirical evaluations of this offset policy to date.

My research is primarily concerned with the operation of biodiversity offsetting as a policy mechanism used to balance losses from development. In this thesis, I examine the application of the EPBC offset policy by reviewing offsets that have been delivered in the Australian Capital Territory (ACT). The majority of these offsets have been required due to ongoing urban expansion and are managed by the ACT Government. This means that many of the influences applicable to offsetting more broadly are not reflected (i.e. clearing due to cropping, changes in state laws), but are exclusively related to urban development and associated infrastructure planned and delivered by the government. Still, pressures leading to habitat loss and modification continue to threaten biodiversity in the ACT (Office of the Commissioner for Sustainability and the Environment, 2015). A key challenge within this jurisdiction is balancing the needs of urban development with environmental protection.

Chapter 2: Literature Review

Biodiversity offsetting involves the protection, and often rehabilitation, of a site in order to compensate for the adverse biodiversity impacts to another (usually associated with development) (Business and Biodiversity Offsets Programme, 2013). Ideally, the conservation gains required through offsetting balance the negative biodiversity impacts (or ‘losses) of development (Peterson *et al.*, 2018). However, there is a lack of empirical evidence on whether offsetting can balance losses from development in practice, and as much of the scientific literature discussed here suggests, balancing losses of biodiversity adequately through offsetting is not theoretically feasible across a broad domain of applications.

It is astounding that so little empirical evaluations of biodiversity offsetting policies exist, given their widespread use, and the significant amount of controversy they attract (e.g. Maron *et al.*, 2016b). Recent literature acknowledges that bridging this gap is a vital challenge to the success of biodiversity offsetting (Bull *et al.*, 2013; Maron *et al.*, 2016b; Carver and Sullivan, 2017).

In this chapter I review the literature on biodiversity offsetting. In particular, I investigate three fundamental principles of biodiversity offsetting policies that aim to facilitate acceptable outcomes for biodiversity. These are: like for like, no net loss and additionally. By exploring studies undertaken to date, current theoretical contentions and gaps in scientific literature are exposed. I then use this literature review to establish the aims of my research.

2.1 Biodiversity offsetting

Biodiversity offsetting is a popular and widely adopted policy instrument. A total of 80 countries have national policies in place, or under development, that mandate biodiversity offsets (Maron *et al.*, 2016b). Biodiversity offsets are mainly delivered by averted loss and/or restoration. Averted loss is where existing threatening processes are removed from the offset site, and restoration is where an offset site is rehabilitated and restored to a higher quality (Figure 1).

Biodiversity offsetting is appealing to governments and the private sector as it enables economic development while supposedly meeting conservation objectives (Bull *et al.*, 2013). Perspectives on offsetting however, both in scientific literature and the public arena, vary substantially, and there is yet to be a consensus as to whether its use in policy should be encouraged (Maron *et al.*, 2016b). Some authors reject the use of offsetting due to the ways that it can reframe nature as a commodity (e.g. Apostolopoulou and Adams, 2017; Spash and Aslaksen, 2015), and the fundamental limitations associated with its implementation (regulatory, political and economic) (Guillet and Semal, 2018). While others propose ways to minimise the risk associated with its use and call for immediate steps to improve policy (Bull *et al.*, 2013; Maron *et al.*, 2016a).

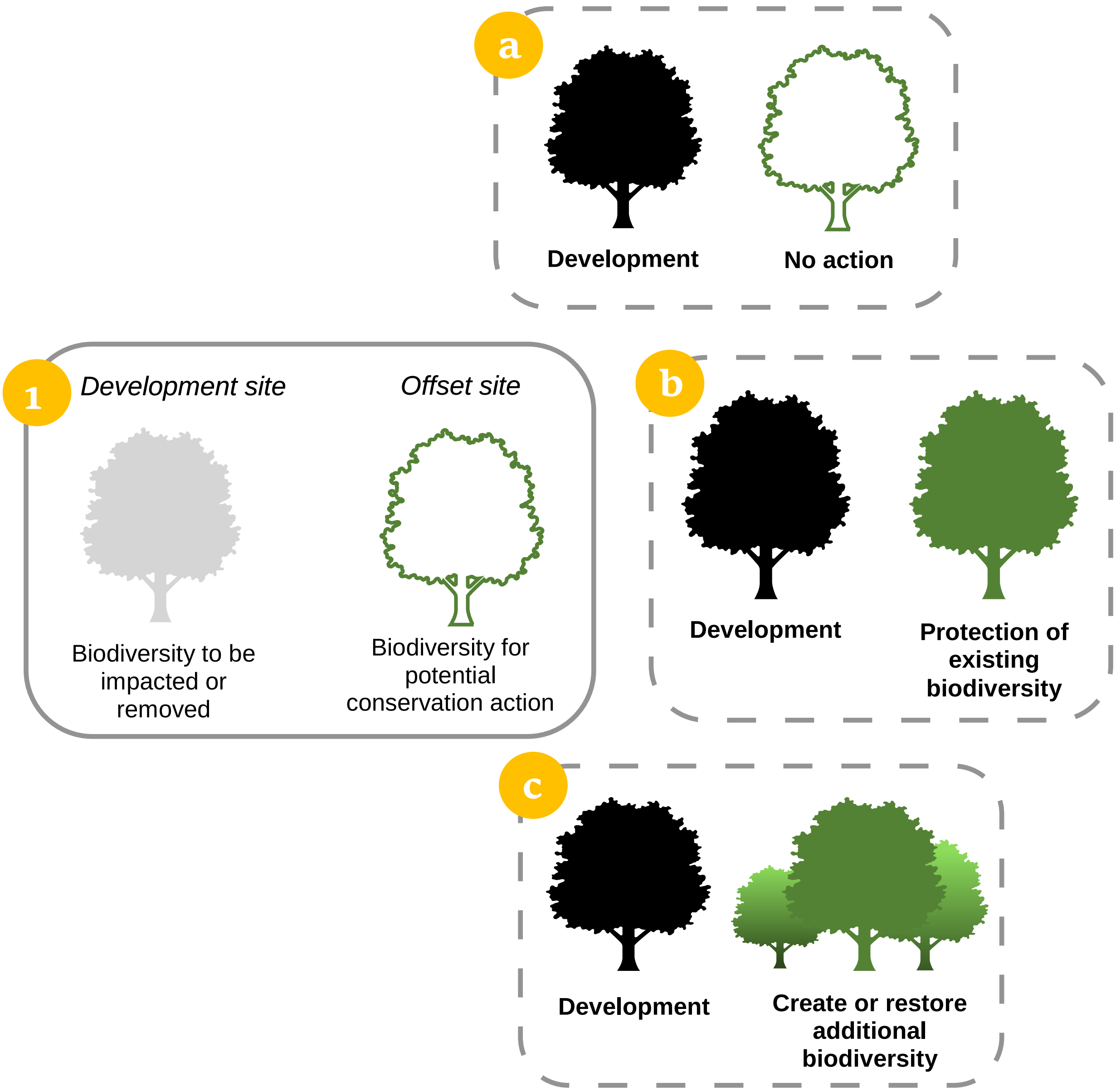

Figure 1. Basic operation of biodiversity offsets, where impacts to biodiversity are contemplated and offsetting is required in order for the development to go ahead (1). Potential options are, development of the site and: (a) no action, resulting in a net loss of biodiversity, (b) actions that stop the decline of existing biodiversity elsewhere (or 'averted loss'), where outcomes reflect the reference scenario and counterfactual applied, and (c) creation or restoration of additional biodiversity elsewhere (diagram based on (Bull *et al.*, 2013)). Note that black shading indicates loss, and green shading indicates gain.

2.2 Principles of biodiversity offsetting

While there are numerous principles utilised in offsetting polices, Bull *et al.* (2013) identify three that are fundamental and common to most policies. These are: like-for-like, no net loss, and additionality. These principles can provide a way of comparing offsetting outcomes at a rudimentary level. The success of an offset policy then could be assessed on the extent to which available data can demonstrate these principles are being met.

As Bull *et al.* (2013) identifies, the various interpretations and assumptions made surrounding these principles is what yields a number of practical and theoretical challenges. Evidence of how each principle has been implemented to date, theoretical perspectives, and key challenges associated with their implementation, are discussed in turn.

2.3 Like-for-like

An offset outcome considered to be 'like-for-like' is one that contains the same type of biodiversity attributes (in type, amount and condition over space and time) to those that have been lost (Maron *et al.*, 2012; Business and Biodiversity Offsets Programme, 2012). This principle is also often referred to as 'in-kind offsets'. In addition to seeking equivalence between the biodiversity attributes exchanged, the like-for-like principle usually aims for equivalency within the landscape context. That is, whether the impact and offset sites differ in their connectivity to other habitats (Business and Biodiversity Offsets Programme, 2012). In this section I review some of the key findings on the implementation of like-for-like outcomes using a case study and a policy evaluation. Then, I discuss achieving 'true' equivalence and an opposing argument to strict like-for-like adherence is discussed in relation to current theoretical literature. Finally, the implications of metric choice in calculating like-for-like are analysed.

2.3.1 Evidence of implementation

Available evidence suggests that biodiversity offsetting is not necessarily achieving gains that are commensurate with losses. For example, recent studies that reflect site and policy level outcomes demonstrate two ways that some of the accounting approaches used for offsets are inhibiting like-for-like outcomes: through simplification and substitution. Carver and Sullivan (2017) use a short-term case study to analyse the biodiversity outcomes of a policy in England and find that habitats are made equivalent in a manner that enables large areas of low biodiversity value to be equal to small areas of high biodiversity value. While an evaluation of a state-level policy in Australia by Gibbons *et al.* (2018) demonstrates that the metrics used are allowing substitution of biodiversity attributes that are more difficult to restore, with those that are relatively easy to restore (for example, mature trees being replaced by establishing tree seedlings). Further, Gibbons *et al.* (2018) identify evidence of substitution at the landscape-scale: losses of habitat (due to development) in more fragmented landscapes tended to be offset with averted losses of habitat occurring in more intact landscapes (Gibbons *et al.*, 2018). Thus, policies that aim for like-for-like offsets may in fact be permitting outcomes that directly oppose the like-for-like objective. This may be exacerbating loss within certain areas, such as highly fragmented ecosystems (Gibbons *et al.*, 2018).

2.3.2 Challenges with implementation

A key flaw of the like-for-like objective is the assumption that two areas, with unique environmental attributes, can be made truly equivalent (Moreno-Mateos *et al.*, 2015), especially since biodiversity is not fungible (Bull *et al.*, 2013). Like-for-like offsets then should support, as close as possible, a combination of the characteristics that are being lost at the impact site. However, an alternative perspective emerging in the literature praises ‘out of kind’ trades, arguing that these may in fact provide greater conservation gains as opposed to strict adherence to the like-for-like objective (Bull *et al.*, 2013; Githiru *et al.*, 2015). The Business and Biodiversity Offsets Programme (2012) also supports this view. They define this aim as ‘like-for-like or better’, where ‘trading up’, as in trading losses in habitat of lesser conservation value for gains in higher value, is favourable.

Metric choice has a large part to play in the outcomes of offsetting policies as there is evidence that the approach chosen strongly influences biodiversity outcomes (Bull *et al.*, 2014), and will invoke varying definitions of what is considered like-for-like. The literature repeatedly warns of the perversities that can arise depending on the approach used to calculate the equivalence (in kind and amount) of biodiversity losses and gains (McCarthy *et al.*, 2004; Gibbons and Lindenmayer, 2007; Maron *et al.*, 2016b). This is because the approach used will determine the attributes ultimately captured by an offset; what is not measured cannot be compensated for (Moreno-Mateos *et al.*, 2015). There are a range of methodologies being used to calculate the gains and losses associated with offsetting, and these have been criticised as either being complex and unclear, or based on simple multipliers (Maron *et al.*, 2016b). The studies discussed earlier (Carver and Sullivan, 2017; Gibbons *et al.*, 2018), highlight the negative outcomes associated with simplified metrics, which evidently contradict the like-for-like objective. This is made possible by allowing individual components of biodiversity to be masked within a single value (Maron *et al.*, 2012), an effect earlier literature warned may occur (Parkes et al., 2003; Gibbons et al., 2009). Thus a key challenge of successfully implementing like-for-like is developing an accounting system that limits simplification and substitution while remaining uncomplicated (Gibbons and Lindenmayer, 2007). This can seem like an impossible task given that biodiversity is innately complex (Bull *et al.*, 2013).

2.4 No net loss

‘No net loss’ is achieved when an offset results in gains that adequately compensate for the losses from development (Gibbons and Lindenmayer, 2007). The aim of no net loss in biodiversity offsetting is usually sought at project level (whether explicitly stated or not), however, over-arching policies (such as the United Nations ‘Land degradation neutrality policy’) seek no net loss on a national or global scale (Maron *et al.*, 2018). This means that, any given offsetting policy will only have the capacity to achieve no net loss in relation to the specific impacts it is designed to account for and is therefore unlikely to be responsible for no net loss of biodiversity as a whole. In this section I review some of the key findings on the implementation of no net loss using empirical evaluation methods and scenario analysis. Then I discuss the implications of choosing a reference scenario and estimating counterfactuals in relation to key theoretical understandings.

2.4.1 Evidence of implementation

While no net loss has been established as possible for individual projects, it is yet to be demonstrated at a policy or program level. To uncover the feasibility of this aim and its associated outcomes, policy level evaluations are required. Authors have repeatedly identified that empirical evaluations of no net loss are scarce (Maron *et al.*, 2010), and the few evaluations available reveal that no net loss has not been achieved (Gibbons *et al.*, 2018). For instance, an evaluation of a habitat compensation policy in Canada reveals that 67% of the projects result in net losses of fish habitat (Quigley and Harper, 2006). In an evaluation of environmental offsets in Western Australia, May *et al.* (2017) estimate that, at most, 39% of the offsets assessed delivered an ‘effective’ outcome. More recently, Gibbons *et al.* (2018) estimate a time delay of 146 years before no net loss of native vegetation will occur under a biodiversity offsetting policy in Australia, an effect that earlier literature warned may arise (Gibbons *et al.*, 2016; Maron *et al.*, 2015; Gordon *et al.*, 2015). Gibbons *et al.* (2018) suggest that this is because biodiversity gains calculated from averted loss offsets can be easily overstated.

There is, however, some evidence that no net loss can be achieved for individual projects (e.g. Norton, 2009; Pickett *et al.*, 2013). Although, these instances are likely restricted to particular circumstances (Gibbons and Lindenmayer, 2007), and a considerable amount of effort is required from both the proponent and government body to achieve no net loss outcomes (Pickett *et al.*, 2013). For example, a case study undertaken by Pickett *et al.* (2013) led to the assertion that no net loss at site level is only achievable where the offset ratio is large (amount impacted versus amount offset), and monitoring is intensive and long-term. It is thus apparent that while no net loss may be theoretically possible at site level, there is very little evidence that this is occurring in practice.

Scenario analysis is an alternative method that can be used to explore the likelihood of achieving policy-level no net loss, and findings from these studies support the growing empirical evidence that no net loss is unachievable under the majority of scenarios. An apparent benefit of this approach is that extensive monitoring data are not required, which could also explain the widespread application of this technique in evaluations of biodiversity offsetting. Sonter *et al.* (2017) and Maron *et al.* (2010) are two key studies that utilise this method and provide commentary on the feasibility of no net loss in practice. Sonter *et al.* (2017) find that no net loss cannot be achieved under any scenarios when they modelled averted loss offsets. Similarly, Maron *et al.* (2010) find that averted loss offsets cannot achieve no net loss of habitat in the case for the red-tailed black cockatoo (*Calyptorhynchus banksii*), rather, such offsets only slow the rate at which habitat is lost. Overall, both empirical evaluations and scenario analyses have indicated that no net loss is unlikely to have been achieved in the majority of applications of biodiversity offsetting policy to date.

2.4.2 Challenges with implementation

The feasibility of achieving no net loss in a theoretical sense has been debated in scientific literature for over a decade. Early concerns and recommendations focused on the importance of dynamic baselines, time discounting, accounting for uncertainty, and efforts in adequate compliance (Gibbons and Lindenmayer, 2007; Bull *et al.*, 2013). Recently, the literature has been substantially focused on the significance of explicitly defining the counterfactual (also referred to as reference scenario) implied by no net loss policies (overarching and impact specific levels), and transparency regarding how these scenarios are calculated (Maron *et al.*, 2018). This shift appears to reflect a growing awareness that a successful no net loss outcome will not necessarily facilitate a greater protection of biodiversity as a whole, and achieving no net loss can have many meanings depending on the counterfactual chosen (Maron *et al.*, 2018).

A counterfactual scenario is applied both at an overarching (policy level), and impact specific levels (site level). A baseline, can be used to inform these scenarios, and is in simple terms is a ‘minimum or starting point used for comparisons’. The use of terminology in relation to counterfactual scenarios is not consistent within the literature, and can be confusing. In this thesis, I use the following:

a) ‘Reference scenario’ is used to describe the policy level counterfactual in general terms, which is an estimated trajectory for biodiversity that is targeted by the policy over time and space (Maron *et al.*, 2018).
b) ‘Baseline’ is used when discussing the specific state or trajectory used to inform the reference scenario/site counterfactual (Maron *et al.*, 2015)
c) ‘Site counterfactual’ hereafter is used solely to describe the scenario calculated at site level which would have occurred had the specific offset not gone ahead.

The policy reviewed here uses the term 'risk of loss' (i.e. the chance that the habitat proposed to be offset will be lost completely due to anthropogenic drivers).

As outlined by Maron *et al.* (2018), there are three broad types of reference scenarios used in no net loss policies (Figure 2), which include: (1) no net loss relative to a fixed scenario (that is, the total amount of biodiversity to be maintained is capped), (2) no net loss relative to a dynamic reference scenario that excludes development (where the impacts targeted by the policy are excluded, and changes through time are considered), and (3) no net loss relative to a dynamic reference scenario that includes development (where changes through time reflect what may have occurred without the introduction of the policy). The second scenario is considered to be the most appropriate for offsetting policies that aim to accomplish no net loss in relation to the impacts that trigger the policy (impact-specific) (Maron *et al.*, 2018).

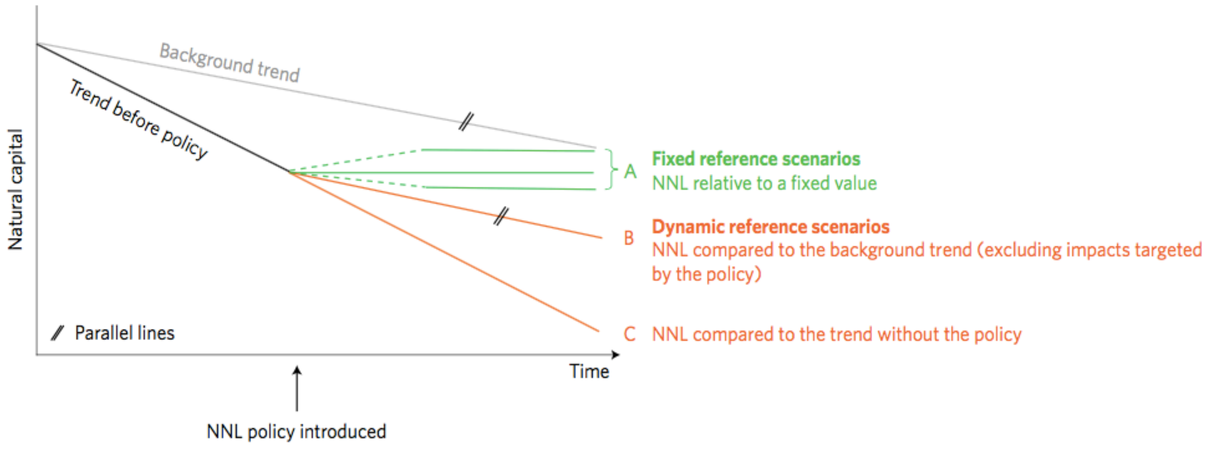

Figure 2. Trends associated with potential reference scenarios of no net loss policies, where natural capital (or biodiversity) is declining through time. Three fixed references scenarios (A), and two dynamic reference scenarios are shown (B and C). The background trend (grey line) is parallel to B as it represents the predicted change in natural capital (or biodiversity) in relation to the impacts targeted by the policy (Maron *et al.*, 2018).

Scenario modelling demonstrates that the reference scenario is a highly influential variable when calculating no net loss (Gordon *et al.*, 2011), hence it is disappointing that a recent review by Maron *et al.* (2018) found that reference scenarios are rarely specified explicitly in not net loss policies. Even more worrying is that all of the baselines used in Australian offsetting policies assume a rate of vegetation loss >5 times steeper than empirical data suggests (Maron *et al.*, 2015), meaning that these crediting baselines severely risk exacerbating biodiversity loss

(Gordon *et al.*, 2015). It is also apparent that the more widespread offsetting becomes under a policy, the more influence the chosen reference scenario will have on biodiversity outcomes more broadly (Gordon *et al.*, 2015; Maron *et al.*, 2018), resulting in an 'entrenching' of the rate of assumed loss implied by the reference scenario (in this case being estimated as far higher than in reality), and masking losses through time (Maron *et al.*, 2018).

Another scenario used to inform offsetting calculations is the site counterfactual (discussed further in section 2.5.2), which also presents serious challenges when attempting to achieve genuine no net loss (Sonter *et al.*, 2017). The site counterfactual is calculated on an offset by offset basis in an attempt to capture what would have happened without the offset going ahead, and thus the loss which is averted by the offset scenario. Some authors recognise that there may be strong incentives to manipulate site counterfactuals so that they are unrealistically steep (i.e. stating that the threats to an offset site are higher than they actually are), which allows an offset scenario to appear as having a higher 'averted loss' than observed in reality (Gordon *et al.*, 2015; Maron *et al.*, 2016b).

2.5 Additionally

The principle of 'additionality' in biodiversity offsetting requires that offsets result in conservation outcomes above and beyond (or additional) what would have occurred if the offset had not taken place (Business and Biodiversity Offsets Programme, 2013). In particular, offsets must be additional to what is required by law, planning regulations or agreed to under other schemes or programs (e.g. Australian Government, 2012a). This principle is fundamental to the success of biodiversity offsetting as a compensatory mechanism. In spite of this, there are no studies that include an assessment of additionality in biodiversity offsetting policies to date, this includes two recent evaluations, namely May *et al.* (2017) and Gibbons *et al.* (2018). In this section I review available commentary on the implementation of additionality and then discuss the negative implications of counterfactual scenarios and cost shifting.

2.5.1 Evidence of implementation

Commentary in the literature suggest that there are serious issues arising with implementing additionality, and indicate that it is likely being undermined by governments in order to appear as meeting national and global targets. A recent article on India's national offsetting program (Narain and Maron, 2018) demonstrates a case where the inadequate implementation of additionality threatens to result in significant environmental consequences. Narain and Maron (2018) identify recent changes in legislation that undermine the purpose of additionality, which will result in a nationwide net loss of biodiversity through 'cost shifting' ('where an agent or organisation, such as a government, seeks to reduce its share of the cost of a service at the expense of another'). Narain and Maron (2018) estimate that the funds being diverted are equivalent to approximately 1.2 million ha of afforestation; this is a significant area of foregone compensation. The passed legislation that allows cost shifting between programmes, blatantly undermines the principle of additionality while allowing the government to meet its global afforestation commitments through funding from the private sector.

The lack of adherence to additionality in biodiversity offsetting to fund protected areas and government-run conservation programs is a growing issue (Maron *et al.*, 2016a). Many protected areas are under-resourced, which leads to poor management, and is especially the case in developing countries (Watson *et al.*, 2014). A consensus in the literature appears to be that this bleak reality somewhat justifies the use of offset finance in delivering conservation goals, particularly where they are unlikely to occur in the near future (Pilgrim and Bennun, 2014; Githiru *et al.*, 2015). However, a number of caveats and safeguards are stressed in order to encourage these endeavours only in specific circumstances: for instance, where gains can be calculated from activities that were not already planned as part of the protected area (Maron *et al.*, 2016a), so that they still meet the requirements of additionality. In practice though, there is speculation that some developed-country jurisdictions are capping or reducing funding to protected areas, with the expectation that ‘offset funding will fill the gap’ (Pilgrim and Bennun, 2014).

2.5.2 Challenges with implementation

Two key ways that the counterfactual can hinder successful implementation of additionality are: through manipulation, and where the counterfactual is not explicitly stated (Maron *et al.*, 2016a). The counterfactual scenario (what would have happened without the offset) is a critical aspect of determining whether an offset meets the principle of additionality. There are a number of uncertainties associated with estimating the counterfactual with respect to future loss or value accrued through time, and as Maron *et al.* (2018) argues, some counterfactuals are more correct than others (say those that are informed by recent trends and use explicit assumptions). Nevertheless, in order for an offset to meet additionality, the counterfactual needs to demonstrate that any of the actions intended by the offset (financial/restorative) were unlikely to have occurred under a ‘business as usual’ scenario. Seeing as the benefits associated with an offset are only relative to the counterfactual scenario, authors rightly suggest that there is an inherent incentive to manipulate it (Maron *et al.*, 2016a; May *et al.*, 2017; Gordon *et al.*, 2015). This may lead to altering the scenario so that it appears worse than in reality (i.e. the threats to the offset site are considered greater and therefore the offset delivers a larger biodiversity gain, which in turn permits greater impacts) (Maron *et al.*, 2016a).

An incentive to cost shift, or ‘wind-back other conservation actions’ is a significant challenge to the successful implementation of additionality (Gordon *et al.*, 2015). Cost shifting is where the benefits created by offsets are used in place of government investment (whether it be planned or potentially forthcoming) (Maron *et al.*, 2016a; Githiru *et al.*, 2015; Gordon *et al.*, 2015). One viewpoint is that an opportunity to contribute to the offset 'market' is diminished when a conservation action is undertaken outside of an offset policy (i.e. land that could have been used as an offset is no longer available, resulting in a lost opportunity) (Gordon *et al.*, 2015). Thus, Gordon *et al.* (2015) argue that governments are incentivised to lower investment in other conservation actions, and even weaken environmental regulations. However, offsets do not function as a market-based mechanism for biodiversity as easily as they do for pollution (Bull *et al.*, 2013), if at all (Koh *et al.*, 2019). It is therefore more likely that governments are incentivised to cost shift or alter previously intended conservation actions due to more prominent stressors, such as lack of funding (Githiru *et al.*, 2015) and pressure to meet targets (Narain and Maron, 2018). Ultimately though, most authors agree that the risk of cost shifting can be minimised by being explicit in policies about additionally, increasing transparency in the offset calculation process and in reporting, especially regarding the counterfactual assumptions used to determine offset gains (Maron *et al.*, 2016a; Maron *et al.*, 2016b; Narain and Maron, 2018).

2.6 The Australian Government's Environmental Offset Policy

From 2004, biodiversity offsetting became the principle policy instrument for regulating land clearing in Australia and most states and territories had formal offsetting arrangements in place by the mid-2010s (Evans, 2016). The Australian Government's environmental offset policy, *The Environment Protection and Biodiversity Conservation Act Environmental Offset Policy* (EPBC offset policy) was introduced in 2012. This policy solely applies to impacts on 'matters of national environmental significance' (MNES), and where 'residual impacts' (or 'unavoidable') are determined 'significant' (as per the *Environment Protection and Biodiversity Conservation Act 1999).* Matters of national environmental significance include (but are not limit to) nationally listed threatened species, ecological communities, and migratory species. Indeed, the Australian Government does not regulate species or ecological communities that are not listed threatened at the national level, and where impacts are not considered 'significant'.

The EPBC offset policy has five 'key aims' and ten 'offset principles' (Table 1). Three of these principles are based around like-for-like, no net loss and additionality. These are: (1) be in proportion to the level of statutory protection that applies to the protected matter, and be of size and scale proportionate to the residual impacts on the matter, (2) deliver an overall conservation outcome that improves or maintains the viability of the aspect of the environment that is protected by national environmental law and affected by the proposed action, (3) be additional to what is already required, determined by law or planning regulations or agreed to under other schemes or programs (Australian Government, 2012a).

The policy seeks to achieve no net loss relative to a dynamic reference scenario (usually declining) of business as usual if neither the impact nor the offset occurred (Figure 2, 'Dynamic reference scenarios' state B) (Maron *et al.*, 2018). It is not clear from the policy or literature to date whether the reference scenarios implied by the policy exclude impacts that are targeted by the policy itself (Maron *et al.*, 2018). The policy states that no net loss is sought in comparison to 'what is likely to have occurred under the status quo, that is, if neither the action nor the offset had taken place'. However, reference scenarios are decided upon on an offset-by-offset basis and the onus is on the proponent to calculate these.

Table 1. Key aims and offset principles of the EPBC offset policy, the numbers in brackets denote the related principle, where (1) is like-for-like, (2) is no net loss, and (3) is additionality.

Key aims	Offset principles
1. ensure the efficient, effective, timely, transparent, proportionate, scientifically robust and reasonable use of offsets under the EPBC Act	1. deliver an overall conservation outcome that improves or maintains the viability of the aspect of the environment that is protected by national environment law and affected by the proposed action **(2)**
2. provide proponents, the community and other stakeholders with greater certainty and guidance on how offsets are determined and when they may be considered under the EPBC Act	2. be built around direct offsets but may include other compensatory measures
3. deliver improved environmental outcomes by consistently applying the policy	3. be in proportion to the level of statutory protection that applies to the protected matter **(1)**
4. outline the appropriate nature and scale of offsets and how they are determined	4. be of a size and scale proportionate to the residual impacts on the protected matter **(1)**
5. provide guidance on acceptable delivery mechanisms for offsets	5. effectively account for and manage the risks of the offset not succeeding
	6. be additional to what is already required, determined by law or planning regulations or agreed to under other schemes or programs **(3)**
	7. be efficient, effective, timely, transparent, scientifically robust and reasonable
	8. have transparent governance arrangements including being able to be readily measured, monitored, audited and enforced
	9. informed by scientifically robust information and incorporate the precautionary principle in the absence of scientific certainty
	10. conducted in a consistent and transparent manner

There are no statistics publicly available on the number of offsets approved under the EPBC offset policy. However, there have been 1,746 actions referred that required approval (defined as ‘controlled action’ decisions) under the EPBC Act since its commencement in 2000. Of these, 106 were decisions made in 2017-18 (Australian Government, 2018b). It is likely that a substantial portion of these assessments (occurring around 2007 onwards) required offsets under the policy introduced here, and in alignment with relevant state offset policies.

A senate inquiry into the EPBC offset policy was undertaken in 2014. One of the key recommendations made by the Committee was that ‘the scheduled five-year review of the *Environment Protection and Biodiversity Conservation Act 1999 Offsets Policy* include consideration and evaluation of the extent to which offsets are achieving positive environmental outcomes’ (Commonwealth of Australia, 2014). Both the scheduled five-year review (as outlined in the policy itself) and an evaluation of the policy’s environmental outcomes have failed to take place.

2.7 Conclusions

My literature review reveals that evaluations of biodiversity offsetting policies that use empirical data are in their infancy. Nevertheless, empirical evaluations of offsetting policies are gradually emerging (Gibbons *et al.*, 2018; May *et al.*, 2017), and these provide valuable insight into the shortcomings of biodiversity offsetting. However, the extent to which offsets are meeting the key principles that underpin biodiversity offsetting policies is not well known. This could be attributed to the lack of suitable data available for thorough evaluations of offsetting policies, which inhibits the ability to evaluate outcomes (Maron *et al.*, 2016b).

It is also evident that there are serious challenges in successfully implementing three key principles of biodiversity offsetting policies: like-for-like, no net loss and additionality. Like-for-like is inherently difficult to achieve, and is further complicated by various metrics used to calculate losses and gains which drastically influence what can be considered like-for-like and the associated biodiversity outcomes (Bull *et al.*, 2014). There is evidence that some metrics enable simplification and substitution of biodiversity attributes at local and landscape scales (Carver and Sullivan, 2017; Gibbons *et al.*, 2018), which may be resulting in outcomes inconsistent with the like-for-like objective. No net loss is a possible (yet hard earned) aim for individual projects (Norton, 2009; Pickett *et al.*, 2013), but is yet to be demonstrated at a policy or programme level. In fact, the majority of projects may result in net losses (Quigley and Harper, 2006) or fail to deliver effective outcomes (May *et al.*, 2017), and large delays in meeting no net loss is also apparent (Gibbons *et al.*, 2018). Finally, while additionality has not been assessed in any biodiversity offsetting policy evaluations to date, there is some evidence of this principle being undermined by governments (Narain and Maron, 2018). This is a topic of growing concern in the literature, particularly in relation to financing protected areas (Pilgrim and Bennun, 2014; Githiru *et al.*, 2015; Maron *et al.*, 2016a).

To date, there have been no formal evaluations of the EPBC offset policy. This is especially surprising given that the EPBC Act is the Australian Government's 'central piece of environmental legislation' which is responsible for conserving vegetation that has already been extensively cleared (Australian Government, 2018a).

2.8 Thesis objectives

In this thesis, I contribute to the gap identified in the scientific literature by providing an empirical evaluation of a biodiversity offsetting policy. Through this evaluation I seek to develop a greater understanding of the extent to which the key principles used in offsetting policies can be achieved. In particular, I aim to evaluate the extent to which the EPBC offset policy had delivered outcomes for biodiversity that:

1. are equivalent or 'like-for-like'
2. have achieved no net loss
3. are additional

Chapter 3: Methods

3.1 Study area

For this research, I reviewed offsets approved in the Australian Capital Territory (ACT) south-eastern Australia (35°16'55.3"S 149°07'42.9"E) between 2010 and 2014. The ACT is home to approximately 420,960 people and has the second fastest growing population in Australia (Australian Bureau of Statistics, 2019). The region encompasses approximately 235,813 hectares (0.03% of Australia's total area): 55% designated to nature conservation, 21% is grazing modified pastures and 13% is urban intensive uses (Australian Bureau of Agricultural and Resource Economics and Sciences, 2018a). Agricultural production is marginal in the ACT, accounting for less than 1% of the national gross value of the agricultural sector (ABRES 2018). The majority of native vegetation clearance is thus due to residential development. This was particularly the case between 2007 and 2011, where 75% of residential development was greenfield (that is, on previously undeveloped sites) while 25% was infill (Office of the Commissioner for Sustainability and the Environment, 2015).

According to the Australian Government's protected matters search tool[1], there are 52 threatened species and three ecological communities that occur within the ACT (protected by the EPBC Act). Two of these ecological communities are critically endangered, (box gum eoodland and natural temperate grassland). The ACT is considered an important area for the protection of box gum woodland as it contains the largest remnants that are in good condition (Threatened Species Scientific Committee, 2006). Natural temperate grassland is seriously under threat in the ACT, as only 5% (1000 ha) of its previous extent remains in moderate to good condition (Environment ACT, 2005).

This study includes all developments and associated offsets listed on the ACT Government's offset register as of December 2018[2] (Figure 3 and Appendix 1) except five mainly due to lack of available documentation (for justification see table at Appendix 2). Indirect offsets, such as funding for research and monitoring were also excluded. In contrast to all other Australian states, the ACT does not have an endorsed, state level offset policy for ACT listed threatened species and ecological communities. As such, offsets in the ACT are required solely under the EPBC Act and are established in accordance with the EPBC offset policy (ACT Government, 2014). This policy used as a framework for delivering offsets for ACT listed species and ecological communities where required (ACT Government, 2014).

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[1] http://environment.gov.au/epbc/protected-matters-search-tool
[2] https://www.planning.act.gov.au/topics/design_build/da_assessment/environmental_assessment/offsets_register

Most of the offsets reviewed here have been approved under the EPBC offset policy which took effect from 2 October 2012. Prior to the release of this policy, the Australian Government was requiring offsets as early as 2001 (Miller *et al.*, 2015). During this time, a draft offset policy (Australian Government, 2007) was used to inform offset calculation and assess suitability (Miller *et al.*, 2015; Gibbons, 2011). While the current policy differs in the offset calculation approach, the main principles are the same. As such, I have included four additional developments and their associated offsets even though they were approved prior to the introduction of the endorsed EPBC offset policy. These are:

1. Molonglo Valley (strategic assessment)
2. Macgregor West 2 Estate
3. Ngunnawal Residential Estate Stage 2C
4. EPIC Block 799 Cabin and Camping development

Of the 10 developments reviewed here, eight are urban development projects (seven of which are considered greenfield projects), the remaining two involved public development projects (Mugga Resource Management Centre expansion and University of Canberra Public Hospital), and all but one of these developments were instigated by the ACT Government (Macgregor West 2 Estate). There are two 'strategic assessments', which are 'landscape scale assessments which consider a much broader set of actions' (i.e. a large urban growth area that will be developed over many years) (Australian Government). The following EPBC listed species and ecological communities required offsetting by these developments:

- Natural temperate grassland of the South-Eastern Highlands (natural temperate grassland) – critically endangered
- White box-tellow box-Blakely's red gum grassy woodland and derived native grassland (box gum woodland) – critically endangered
- Golden sun moth (*Synemon plana*) – critically endangered
- Pink-tailed worm-lizard (*Aprasia parapulchella*) - vulnerable
- Striped legless lizard (*Delma impar*) – vulnerable

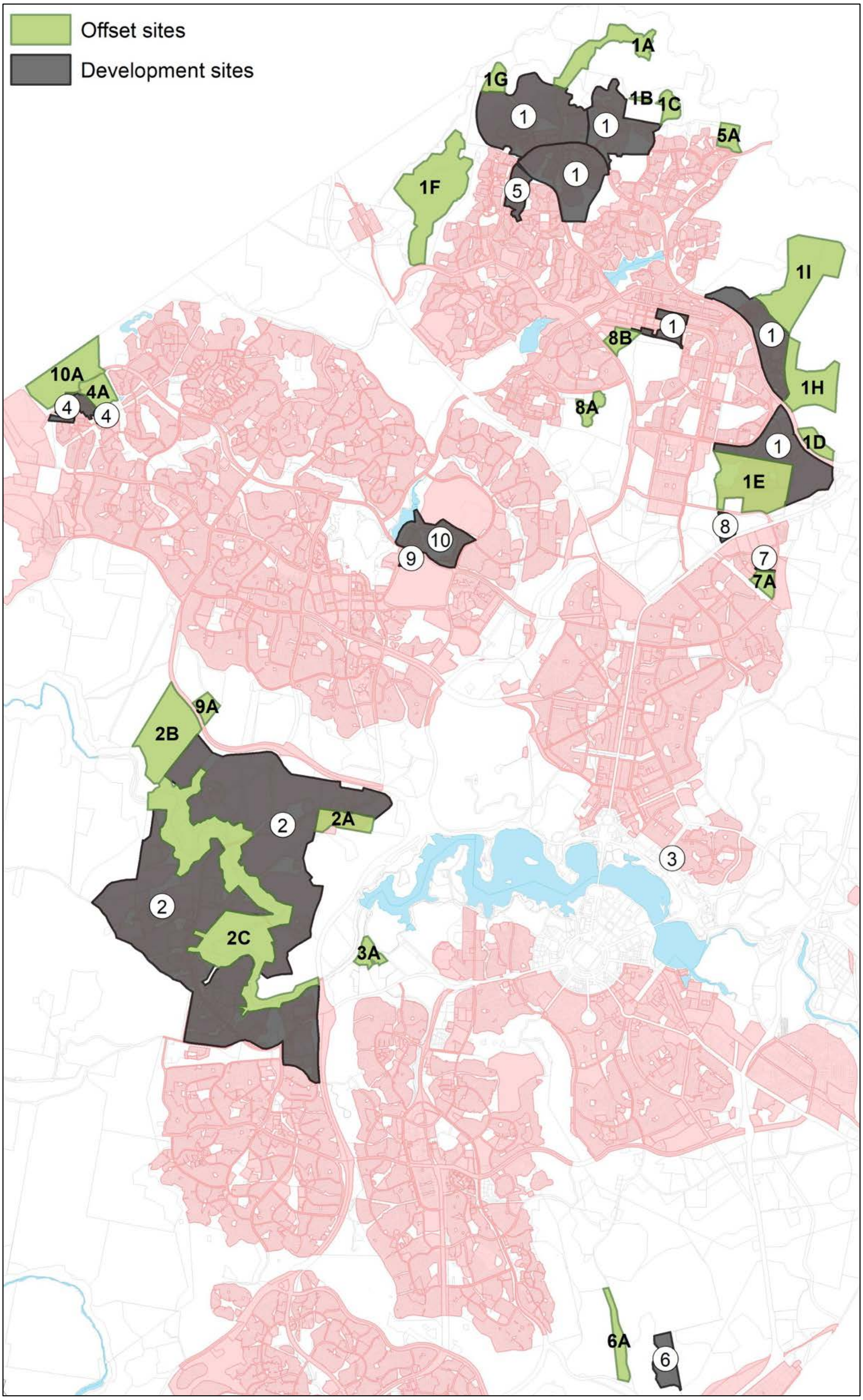

Figure 3. Development sites (1-10) and corresponding offset sites (1A-10A) in the ACT (see Appendix 1 for list of names), created using Arcmap.

3.2 Data collection

Data on development and offset sites were obtained primarily through a review of documents contained in the ACT Government's offset register[3] and the Australian Government's public notices website[4]. These documents included environmental impact statements, offset plans, approvals and recommendation reports for five of the 10 developments included in this review. From these sources, the following data were captured:

a) Approval year: when the development was approved by the Australian Government.
b) Proponent: the entity responsible for the development and offset.
c) Impact area: the threatened vegetation directly impacted by development and assessed as a 'significant impact', and further categorised as an ecological community or habitat for threatened species (where overlapping, precedent was given to the ecological community classification).
d) Offset area: the area directly offset, and further categorised by ecological community or habitat for threatened species.

Where possible, the size of the final impact area and offset area have been derived from approval conditions and compliance reports, as these are considered to provide the most up to date and accurate data.

I sourced spatial data from the ACT Government's geospatial data catalogue[5]. The datasets utilised from this source are summarised in Table 2, and are all polygon data. All development sites (except for the two 'strategic assessments') were manually geocoded using Arcmap with reference to maps available through the Australian Government's public notices website and using satellite imagery. I then combined these with the 'Strategic Assessment Areas' dataset.

I also used the Australian Government's Collaborative Australian Protected Area Database[6] (CAPAD) for figures on the total area protected in nature reserves in the ACT through time (2002 to 2016, reported ever 2 years), and accessed historical versions of ACT's Territory Plan (2008) using the ACT Legislation Register[7] to identify the zoning of each offset site at the time of approval.

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[3] https://www.planning.act.gov.au/topics/design_build/da_assessment/environmental_assessment/offsets_register
[4] http://epbcnotices.environment.gov.au/referralslist/
[5] https://actmapi-actgov.opendata.arcgis.com
[6] http://www.environment.gov.au/land/nrs/science/capad
[7] https://www.legislation.act.gov.au/ni/2008-27/

Table 2. Description of datasets (all versions used were last updated on 12/11/2018).

Dataset	Description
Environmental Offsets	Indicative of environmental offsets in the ACT
Strategic Assessment Areas	Indicative of strategic assessment (development) boundaries
ACT Vegetation Map 2018	Most recent vegetation mapping for the ACT
Threatened Woodland	Distribution of ACT and EPBC listed Box Gum Grassy Woodland in the ACT
Threatened Fauna Habitat	Areas of known habitat for terrestrial mammals, reptiles, birds and invertebrate listed as threatened in the ACT

3.3 Analyses

3.3.1 Like-for-like

To evaluate whether the offsets met the principle of like-for-like I compared the following at development sites and offset sites:

1. Total area, and the ratio of area offset to area developed (offset ratio) (using data acquired from assessment and approval documentation).
2. The proportion of each vegetation type represented, including a delineation of the extent of native versus exotic vegetation (using the ACT Vegetation Map 2018 and overlay analysis tools in ArcMap) (see Appendix 3 for native vegetation classification).
3. Fragmentation in the broader landscape and the potential implications for species richness (using data from the previous analysis).

To evaluate the proportion of each development and offset site occupied by different vegetation types and the proportion of each site that was native or exotic vegetation, I used data from the ACT Vegetation Map 2018. These data required several manual corrections once exported. This is because in some cases clearing (associated with the approved development sites) had preceded aerial imagery used to generate the layer. Therefore, the native vegetation occurring at these sites before development was not represented. An example is provided at Appendix 5. In any other cases where the spatial data diverged from that provided in documentation, no modifications were made.

To evaluate fragmentation, I calculated the percentage of native vegetation occurring around development and offset sites. I applied 200m, 500m and 1km buffers around each site in ArcMap.

Using the ACT Vegetation Map 2018, I identified the vegetation types falling within these buffers. These vegetation types were categorised as either native or non-native (see table at Appendix 3 for a list and explanation of these categories). I then calculated the percentage of native vegetation that occurs around each site at the three scales (200m, 500m and 1km buffers). I applied two-sample t-tests assuming equal variances to test whether the mean percentage of native vegetation differed significantly between the development sites and offset sites for each buffer (200m, 500m and 1km).

I used data on the percentage of native vegetation around each site to test whether there would be a net loss in species richness as a result of future hypothetical development. To do this I used methods similar to those outlined in Fischer and Lindenmayer (2007). The cover of native vegetation at broad scales has been used as a surrogate for predicting species richness in fragmented landscapes (Radford *et al.*, 2005; Cunningham *et al.*, 2014). So, I used the % cover of native vegetation around development and offset sites (separately, using data from the 200m, 500m and 1km buffers described previously) to predict the proportional change in species richness using a generic species-area curve ($A^{0.25}$). Where A is the area of native vegetation and 0.25 is a generic exponent employed in fragmented landscapes (Preston, 1962; Connor and McCoy, 1979). I predicted the loss in species richness across all development sites based on the % cover of native vegetation before and after development and compared this with the gain in species richness before and after the implementation of offsets. I then summed losses and gains to determine the net outcome for species richness. These analyses were undertaken separately for the three buffer distances.

Using the results for the 200m buffer (% of native vegetation surrounding development sites and offset sites), I then modelled the trend in % change of species richness at hypothetical development sites and offset sites for every additional 1% developed and offset. The 200m buffer was used because the results from the previous test demonstrated that all three buffer sizes were similar in the estimated proportion of surrounding native vegetation. From this, I determined the additional % gain in habitat required at offset sites in order to fully compensate for the losses occurring at development sites.

3.3.2 No net loss

To determine whether offsets implemented in the ACT achieved no net loss I estimated the delay in years before there would be no net loss in the area threated native vegetation. This analysis was based on a modified version of the methods employed by Gibbons et al. (2018), which involved using the net area of threatened vegetation (that is, threatened ecological communities and habitat for threatened species) lost at the development sites and the net area gained at the offset sites (as reported in assessment and approval documentation). Comparable methods have also been applied in a number of simulation-based studies (e.g. Sonter *et al.*, 2017; Gordon *et al.*, 2011).

Ideally, the calculated gains associated with each offset site would be delineated into restoration and averted loss. That is, the proportion of overall gains associated with the offset that relate to restoration, and those that are representative of averted loss. However, the methods and inputs for the gains/losses calculation for offsets are not publicly available. No data was available that indicate the size of averted loss gains, or how this was calculated (e.g., counterfactual scenario/crediting baseline). Gibbons et al. (2018) used the extent of the canopy cover of woody vegetation (%) (at the time the offset was established) to classify gains as restoration and/or averted loss. A key limitation of this method is that it can only represent the extent of woody-vegetation types, and as such, non-woody habitats such as grasslands were not represented in their study. This was not an option for this project, given that grasslands constitute a substantial proportion of the vegetation impacted (63%) and offset (49%) in the ACT (not including grassy woodland formations). Instead, I estimated the proportion of offset area that was averted loss and restoration by simulating scenarios ranging from 75% restoration and 25% averted loss to 25% restoration and 75% averted loss (Table 2). I discuss the implications of this assumption at Appendix 4.

Averted loss gains require an estimation of the reference scenario and counterfactual (what would have happened to the proposed offset site had it not been secured). I calculated gains from averted loss over *t* years as:

(1)

$$\sum_{t=1}^{n}(O \times P \times r)$$

where *O* is the total area offset at *t* years, *P* is the proportion of the offset area that is averted loss, and *r* is the annual rate of loss of the impacted vegetation under the reference scenario. The policy reviewed here does not explicitly state a reference scenario: instead, the onus is on the proponent to estimate and explicitly state the scenario in which gains are calculated (Maron *et al.*, 2015). Details regarding the reference scenario, or site counterfactual used for each offset was not available. In lieu of these data, Gibbons et al. (2018) quantified the annual loss of woody vegetation across their study area and used this as the baseline (across all offsets analysed). Similarly, in a review of Australian offset policies, Maron *et al.* (2015) used recent rates of woody vegetation loss as the baseline to inform their analysis. While the EPBC offset policy calculator incorporates both the risk of loss of extent and risk of condition decline, it was only possible to estimate the risk of loss related to extent because data on the risk of loss related to condition was not available. Since only a small amount of natural temperate grassland (5.5 ha) was offset, while a large amount of box gum woodland was offset (727.4 ha), I decided to use the extent of box gum woodland to inform the reference scenario (*r*). I estimated the annual loss of box gum woodland in the ACT as 0.004%. To do this I used available figures on the total area of box gum woodland occurring pre-1750 within the ACT (32,000 ha) and the most current estimate published by the Australian Government (10,865 ha) (Threatened Species Scientific Committee, 2006). I calculated annual loss using the following formula:

(2)

$$(F/S)^{1/y}-1$$

where F is the estimated current extent of box gum woodland (10,865 ha), S is the extent of box gum woodland prior to clearing (32,000 ha), and y is the years that have lapsed between these estimates (256 years). I used this estimated risk of loss (0.004%), and the equation used to calculate averted losses (described earlier) in order to estimate gains associated with averted losses. Then, gains associated with restoration were calculated as ($O \times S \times Pr$) (3). Where O is the total area offset at t years, S is the success rate of restoration and Pr is the proportion of the offset area that is restoration. Keeping in mind that different types of restoration are more likely to be successful than others (Maron *et al.*, 2012), I applied restoration success rates ranging from 10% to 100% to capture 'worst case' to 'best case' success rates. This is because there are multiple variables that affect restoration success, and these can differ across timeframes (Maron *et al.*, 2012). For simplicity, I simulated restoration success rates assuming restoration will yield outcomes (or gains) at t years. While this is clearly a simplification, it was not within the scope of this project to test different annual success rates. See Appendix 4 for a statement of the key assumptions and how they are likely to affect my analysis.

Given the uncertainty associated with the proportion of offsets that were averted losses and restoration, I investigated the length of time (years) before no net loss of threatened native vegetation would be achieved under three different scenarios. These varied in terms of: (a) the percent of gain obtained by restoration, (b) the percent of gain obtained by averted loss, and (c) the restoration success rate. These scenarios are summarised in Table 3. The EPBC offset policy permits the calculation of averted losses over a maximum of 20 years, and places higher value on offsets that deliver restoration gains over a shorter time period. Thus, these scenarios were assessed in relation to a 20-year timeframe. It is recognised that restoration projects may have timeframes that extending beyond this 20-year period.

Table 3. No net loss scenarios simulated.

Scenario	Per cent of offsets based on restoration	Per cent of offsets based on averted loss	Restoration success rate	Risk of loss (baseline) per annum
1	75%	25%	10-100%	0.004%
2	50%	50%	10-100%	0.004%
3	25%	75%	10-100%	0.004%

Then, I tested the impact that the estimated risk of loss (or baseline) applied has on the estimated years before no net loss. To do this, I used both the annual rate of loss implied by the ACT offset policy (yet to be ratified) of 0.5% (Maron *et al.*, 2015), and my own background rate of loss estimate of 0.004%. Using these I compared the years before no net loss would be achieved (assuming all area offset was 100% averted gains and 0% restoration). Maron *et al.* (2015) excluded the ACT from their baseline comparison because the authors were unable to detect vegetation change by satellite imagery.

3.3.3 Additionality

I addressed my third research question (i.e. have offsets met the principle of additionality) by first analysing whether the long-term rate of formal reservation in the ACT changed after the introduction of biodiversity offsetting (2012). I hypothesised that, if offsets were additional then the rate of formal reservation in the ACT should remain unchanged. To do this, I summarised the total area (ha) gazetted as Nature Reserve in the ACT by using CAPAD data spanning 2005 to 2014. I calculated the additional area (ha) gazetted as Nature Reserve in the ACT for five years before (2005-2009) and five years after (2010-2014) offsetting was introduced and the proportion of this that was designated as biodiversity offsets.

Secondly, I analysed whether financing for conservation responsibilities has changed since the introduction of offsets in the ACT. Using the financial data available in compliance reports on the offset register, I summarised the total expenditure allocated to the management of offsets by year. I also used budget statements available on the ACT Governments website to review the estimated outcome per year for Conservation and Land Management in the ACT. Using both of these, I intended to compare the expenditure for Nature Reserves with respect to that allocated to offsetting, before and after the introduction of offsetting. I hypothesised that, if offsets were additional then the trend in offsetting expenditure (likely increasing through time) would not be reflected in the expenditure allocated to Nature Reserves and other conservation land management in the ACT (or an increase to reflect any additional reserves).

Lastly, I evaluated whether there are cases of offsets being delivered that overlap with existing statutory obligations (i.e., in legislation or instruments such as plans or agreements developed under legislation). To do this, the zoning for each offset site at the date of approval were identified in the relevant version of the ACT’s Territory Plan (2008). Where offset sites extended over more than one land-use zone, I used the zone that represented the greatest overall proportion of the offset area (if there were shapefiles available for previous versions of the Territory Plan, this would have been more accurate).

Chapter 4: Results

4.1 Summary statistics

A total of 10 developments, two of these being strategic assessments (i.e. Molonglo Valley and Gungahlin), were included in this study. These developments were approved between 2010 and 2014. Eight of these developments are urban expansion projects and the remaining two involved public development projects. I calculated a total of 567 ha of threatened vegetation (10 development proposals) approved for clearing subject to the establishment of 1,328 ha of biodiversity offsets (21 offset sites). Figure 4 summarises the amount of threatened vegetation impacted by development and that set aside for offsets per year.

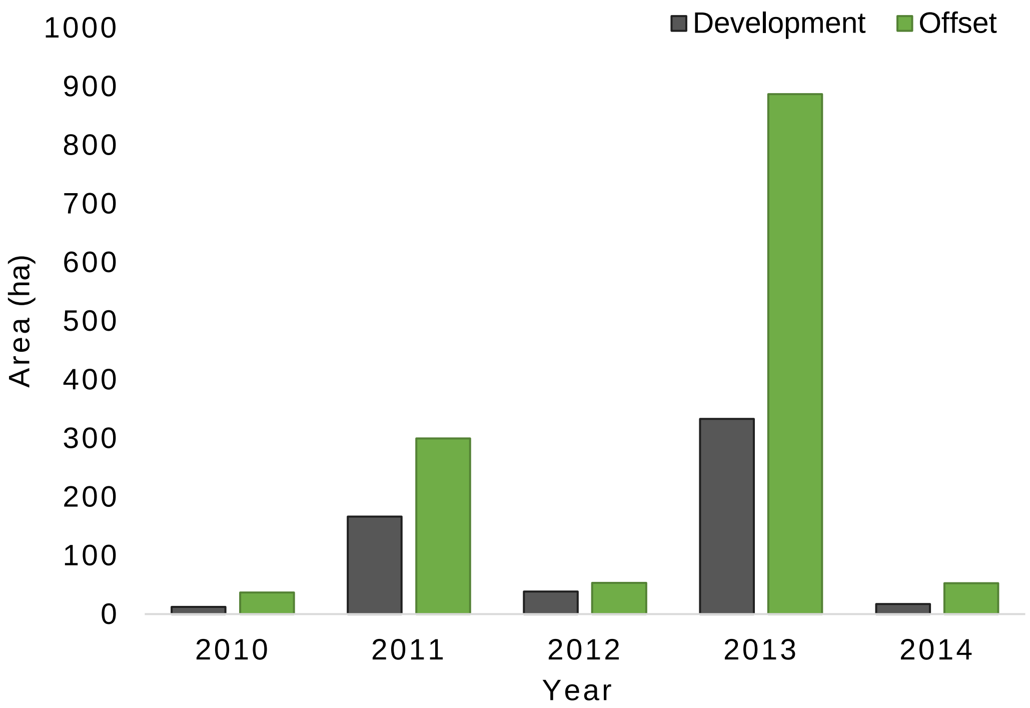


Figure 4. Total area impacted by developed and offset by year in the ACT by year.

I compared the areas provided on the ACT Offset Register and EPBC Referrals List (Table 2) with that derived from spatial data in ArcMap, and found a negligible difference between these sources (Figure 5). This demonstrates that the development sites shapefile (that were manually geocoded) are fit for purpose. I delineate which data type (spatial data or documentation) was used for the subsequent analyses.

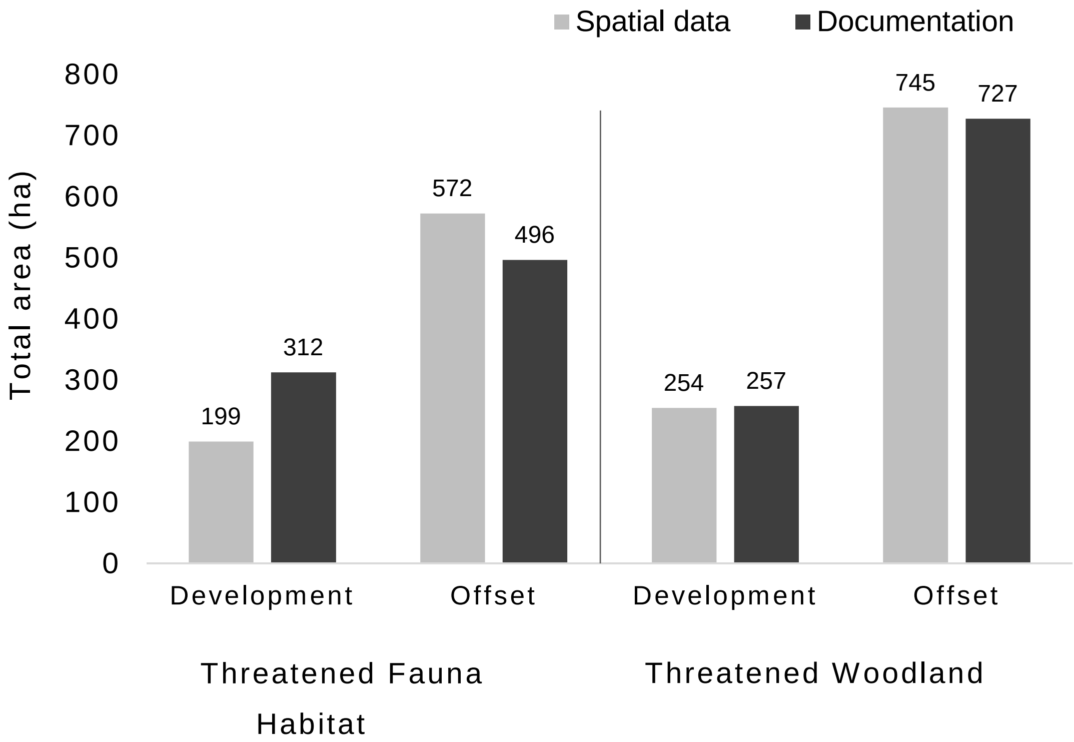


Figure 5. Comparison of totals (ha) provided in assessment and approval documentation and that sourced through spatial data (rounded) for Threatened Fauna Habitat and Threatened Woodland (box gum woodland) and site type (development and offset).

4.2 Like-for-like

I evaluated like-for-like between development and offset sites in terms of: vegetation type, vegetation condition, fragmentation in the broader landscape and potential implications for species richness.

4.2.1 Vegetation type

From the documentation reviewed, I classified the vegetation impacted by development and that offset, see figures at Table 4 (for exact figures by development see Appendix 6). Overall, I calculated an offsetting ratio of approximately 2.3 ha for every hectare of threatened native vegetation approved for clearing. Ecological communities (box gum woodland and natural temperate grassland) had a higher offsetting ratio, at 2.8 ha offset for every hectare approved for clearing. While habitat for threatened species (including: golden sun moth, striped legless lizard and pink-tailed worm-lizard) had a lower offset ratio overall, at 1.6 ha offset for every hectare approved for clearing.

Table 4. Development and offset site vegetation characteristics and offset ratios (derived from assessment and approval documentation).

Classification	Development sites (ha)	Offset sites (ha)	Ratio
Ecological communities	259.2	732.9	2.8
Box gum woodland	257.4	727.4	2.8
Natural temperate grassland	1.8	5.5	3.1
Habitat for MNES*	312.5	496	1.6
Native vegetation (other)	-	171	-
Threatened vegetation (total)	567.2	1328.4	2.3

*Matters of national environmental significance, or 'threatened species'

Using the ACT Vegetation Map (2018), I found that development sites included a greater proportion of exotic vegetation (20%) than that at offset sites (7%) (Figure 6). For totals (ha), see Appendix 7.

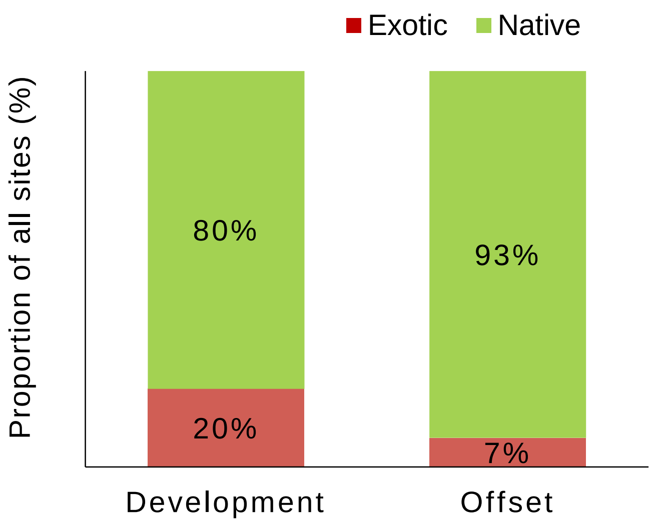


Figure 6. Proportion of exotic and native vegetation as a % of the total (vegetated) area within development sites (1953.8 ha) and offset sites (1617.7 ha) (derived using spatial data).

Using the same dataset, I found that, overall native grasslands made up the majority of land approved for clearing (68%), while encompassing a smaller per cent of the total area offset (45%) (Figure 7). The only other vegetation type with a greater per cent of total area represented in development sites were exotic vegetation types (grassland, shrub-land, woodland and forest inclusive). All other vegetation types had a greater representation in offset sites compared with development sites. For totals (ha) see Appendix 7.

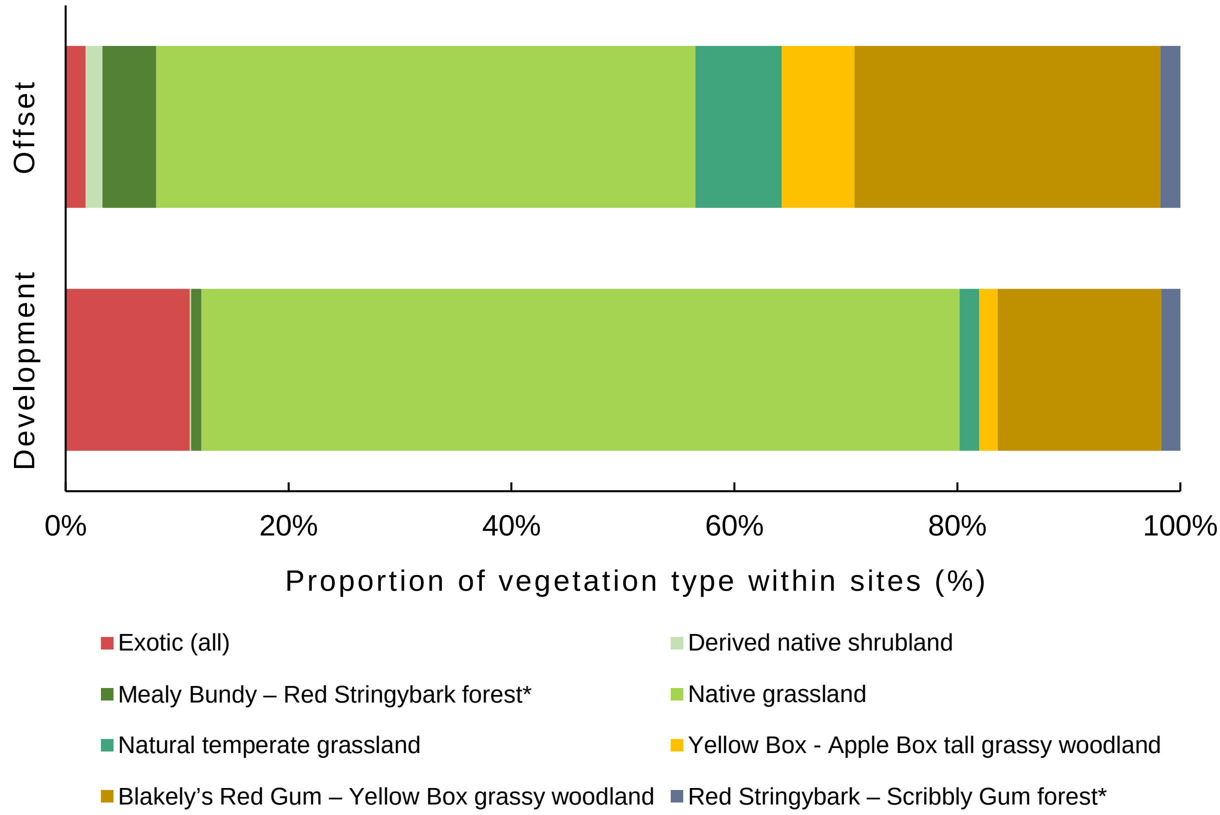


Figure 7. Vegetation types as a % of the total (vegetated) area within development sites and offset sites (derived using most recent spatial data) (aquatic fringing vegetation removed due to values below 0%).

4.2.2 Fragmentation and species richness

Using the ACT Vegetation Map (2018) I estimated mean proportion of native vegetation (%) occurring around development sites and offset sites at using three buffer sizes (200m 500m, 1km). Across all buffers, offset sites had a higher proportion of native vegetation than that at development sites (Figure 8). I found that the mean proportion of native vegetation occurring around offset sites was significantly ($p<0.05$) different to that occurring around development sites at all three buffer sizes (200m, 500m and 1km) (Table 5). That is, on average, development sites contained significantly less native vegetation cover in the surrounding landscapes than offset sites at all buffer sizes. The mean proportion of native vegetation (%) around offset sites decreases marginally as the buffer size increases.

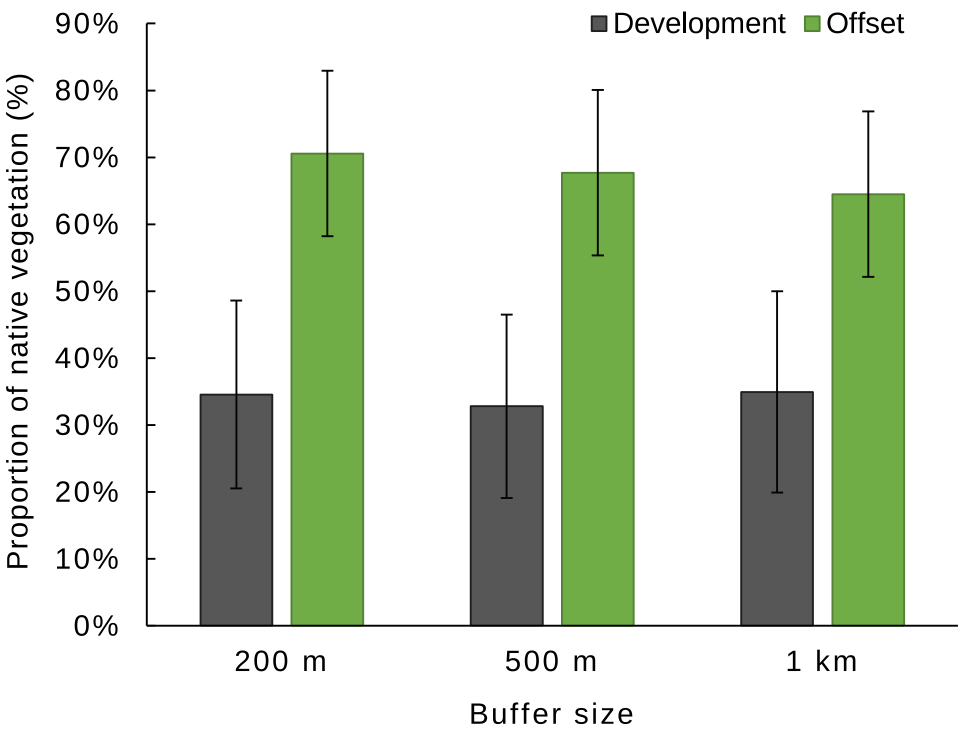


Figure 8. Mean per cent (± 95% confidence intervals) of native vegetation within 200m, 500m and 1km buffers of development and offset sites.

Table 5. Tests of significance of the differences in mean per cent of native vegetation around development sites (n=10, df=9) versus offset sites (n=21, df=20) based on two-tailed t-tests.

Buffer (m)	Site type	Mean (%)	Standard error (%)	Prob>[t]
200	Development	35	7.02	0.0014
	Offset	71	6.18	
500	Development	33	6.86	0.0018
	Offset	68	6.19	
1000	Development	35	7.52	0.0069
	Offset	65	6.01	

I used these findings to test whether there would be a net loss in species richness as a result of development by estimating the proportional change of species richness at development sites and offset sites using the species area curve (Lindenmayer and Fischer, 2006). I predicted an overall loss in species richness when considering incremental losses in native vegetation, and gains associated with the proportion of native vegetation surrounding development and offset sites (Table 6). Using the 200m buffer results for the mean % of native vegetation around development sites (35%) and offset sites (71%) and assuming that species richness can be predicted with the area of habitat raised to an exponent of 0.25, I found that in order to achieve a neutral outcome in species richness, offsets would need to increase the % of habitat available by at least double (Figure 9).

Table 6. Estimated overall net loss or gain in species richness (%), by calculating the loss and gain (%) of species richness associated with 1% of additional vegetation developed and offset.

Buffer (m)	Predicted % loss of species richness with every 1% of landscape developed	Predicted % gain of species richness with every 1% of landscape offset	Net loss in species richness (%)
200	-0.56	0.32	-0.24
500	-0.58	0.33	-0.25
1000	-0.56	0.35	-0.21

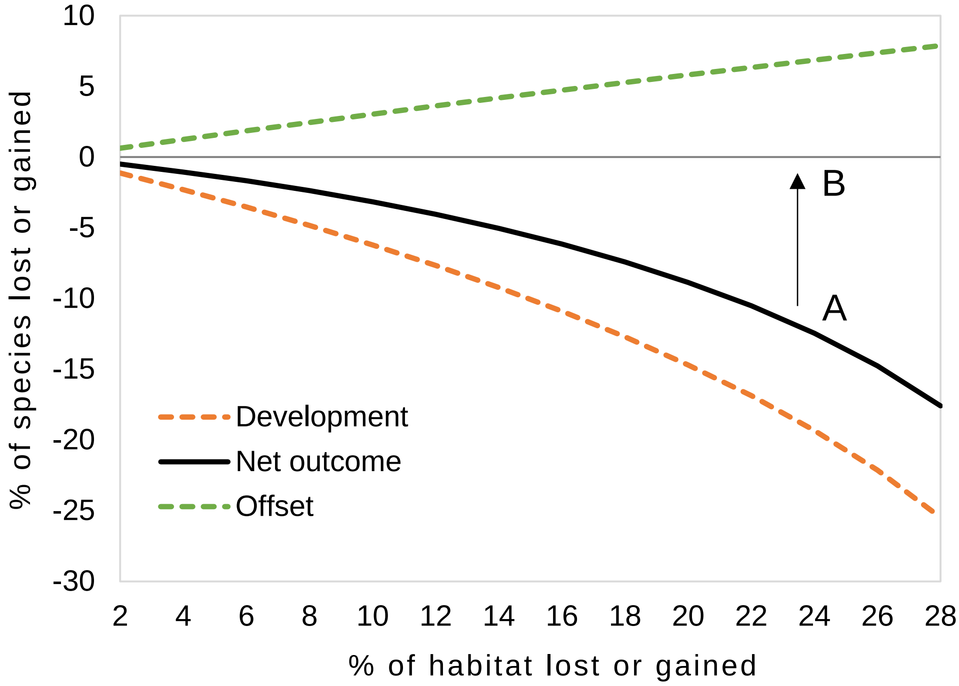


Figure 9. Predicted loss in species richness in the study area with every % of native vegetation cleared for development (dashed orange line) and the

predicted gain in species richness (dashed green curve) with every % of habitat offset. Predictions are based on species-area curve in which area is raised to an exponent of 0.25 (Fischer and Lindenmayer, 2007). The net outcome from development and offsetting is demonstrated by the black solid line. In order for a neutral outcome to be achieved (move net outcome from A to B), offsets would need to (at least) double the % habitat gained at offset sites for every % lost at development sites.

4.3 No net loss

Using scenarios that varied according to the % offsets that included averted loss gains and restoration gains (Table 3), and the success rate of restoration, I estimated in how many years no net loss could be achieved. The period before no net loss was achieved ranged between 1 and 356 years (Figure 10) depending the scenario, and using a crediting baseline of 0.004% per annum. The conditions in which the policy is most likely to deliver no net loss within the 20-year timeframe specified in the EPBC offset policy, are those below the dashed line in Figure 10.

The following are the scenarios that are able to reach no net loss within the 20 year time frame:

1. Gains are associated with 75% restoration and 25% averted loss (scenario 1), and a restoration success rate that is ≥60% within 20 years.
2. Gains are associated with 50% restoration and 50% averted loss (scenario 2), and a restoration success rate that is ≥80% within 20 years.

Because gains associated with averted loss take a long time to accrue, and the higher proportion of restoration gains the greater the impact of the restoration success rate, scenario 3 (25% restoration, 75% averted loss) cannot reach no net loss within the 20-year time frame even if restoration was 100% successful. That is, based on my assumptions, an offset strategy that is predominately averted loss will not achieve no net loss within the 20-year timeframe specified in the EPBC offset policy. A strategy based solely on restoration offsets could theoretically achieve no net loss if the success of restoration 20 years was ≥43%.

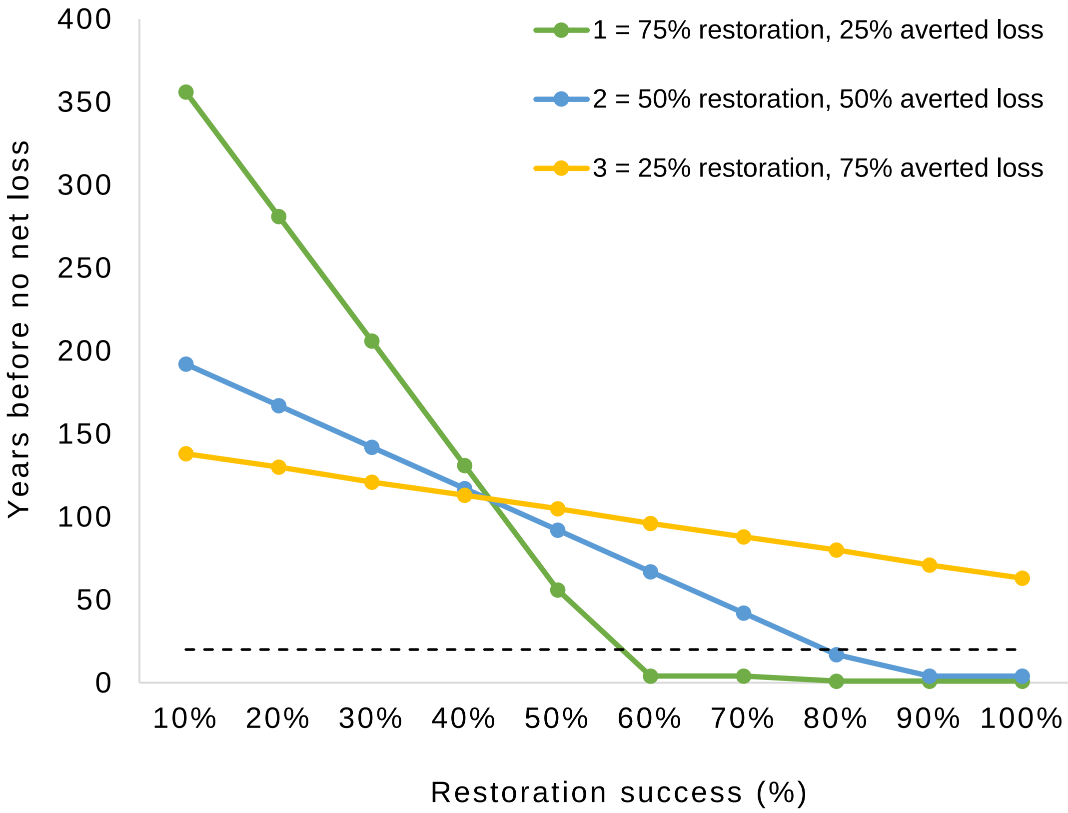


Figure 10. The predicted years before no net loss (in terms of the area of threatened vegetation) will be achieved based on scenarios representing varying success rates for ecological restoration and varying percentages of averted loss at offset sites. All scenarios are based on an assumed background rate of loss of 0.004% per annum. The dashed line is the maximum time over which averted losses can be calculated under the policy examined here (20 years).

I compared two rates of loss (assuming that 100% of the offsets were based on averted loss), and calculated the following delays before no net loss would be achieved:

1. Using 0.004% per annum, no net loss would be achieved in 111 years.
2. Using 0.5% per annum, no net loss would be achieved in 5 years.

4.4 Additionality

I analysed the rate of formal reservation in the ACT through time, and compared this with the area gained through offsetting (Figure 11). A total of 1,495 ha was set aside for reservation in the five years before biodiversity offsetting was introduced (2005-2009), while 446 ha was reserved in the five years after biodiversity offsetting was introduced (2010-2014) (excludes 'offsets that have been formally reserved') (Figure 11). A total of 308 ha was formally reserved in 2014 which is comprised of four offsets (reviewed here), namely: Gungaderra Grasslands

(22 ha), Isaacs Ridge (37 ha), Kinlyside (228 ha) and Mulanggari Grasslands (21 ha). Totals are provided at Appendix 8.

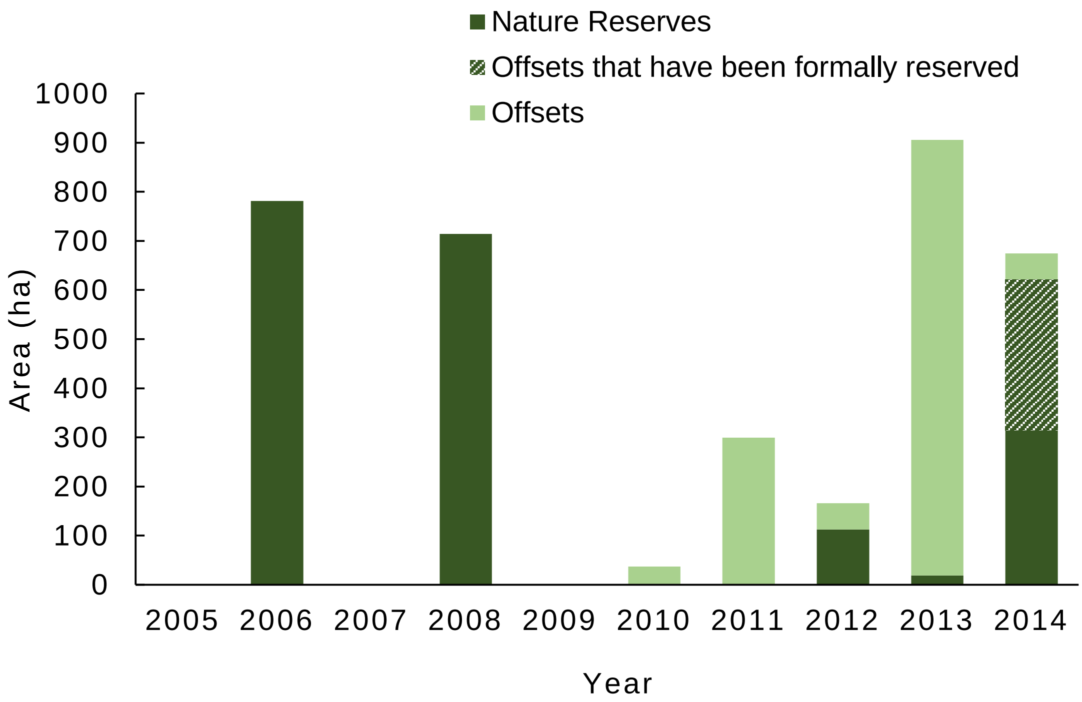


Figure 11. Additional land (ha) allocated by year for offset sites (n=21) and Nature Reserves in the ACT (n=44) (4 of which are also offsets) from 2005 to 2014, where offsetting was introduced informally in 2010 and the EPBC offset policy finalised by 2012.

I have provided a summary of the total land offset by land tenure (at the time of approval) in Table 7. The majority of land set aside for offsetting was zoned as suburban at the time of approval (42%), which is largely associated with offsets provided for the Gungahlin district development (681 ha) (Figure 12). The tenure with the next highest representation within offsets was river corridor (27%), which was entirely associated with a particular offset provided for the Molonglo Valley development.

Table 7. Proportion of total area offset (%) by land tenure (greatest amount occupied where multiple zones) based on relevant version of the ACT's Territory Plan at the time of the offset approval under the EPBC Act.

Tenure	Description (from 'zone objectives' stated in Territory Plan)	Proportion of total area offset*	Area* (ha)
Suburban	Low rise and low density residential areas where housing is predominately single dwelling.	42.47%	747.27
River corridor	Protects stream flow, water quality and flood plains from adverse impacts, and conserves the ecological and cultural values of major river corridors.	26.66%	469.09
Hills ridges and buffers	Conserves the significant cultural and natural heritage resource and diversity of natural habitats and wildlife corridors.	18.61%	327.50
Broadacre	Predominately in rural landscape for uses which require larger sites and/or distance from urban areas.	7.72%	135.87
Urban open space	Supports recreational activities and the protection of flora and fauna habitats, corridors, and natural and cultural features.	3.24%	56.98
Open space (Designated area precinct)	Governed by the National Capital Authority, this zone contains areas identified as having special characteristics of the National Capital (i.e. Parliamentary Zone, Lake Burley Griffin).	1.29%	22.64

* using ArcMap data

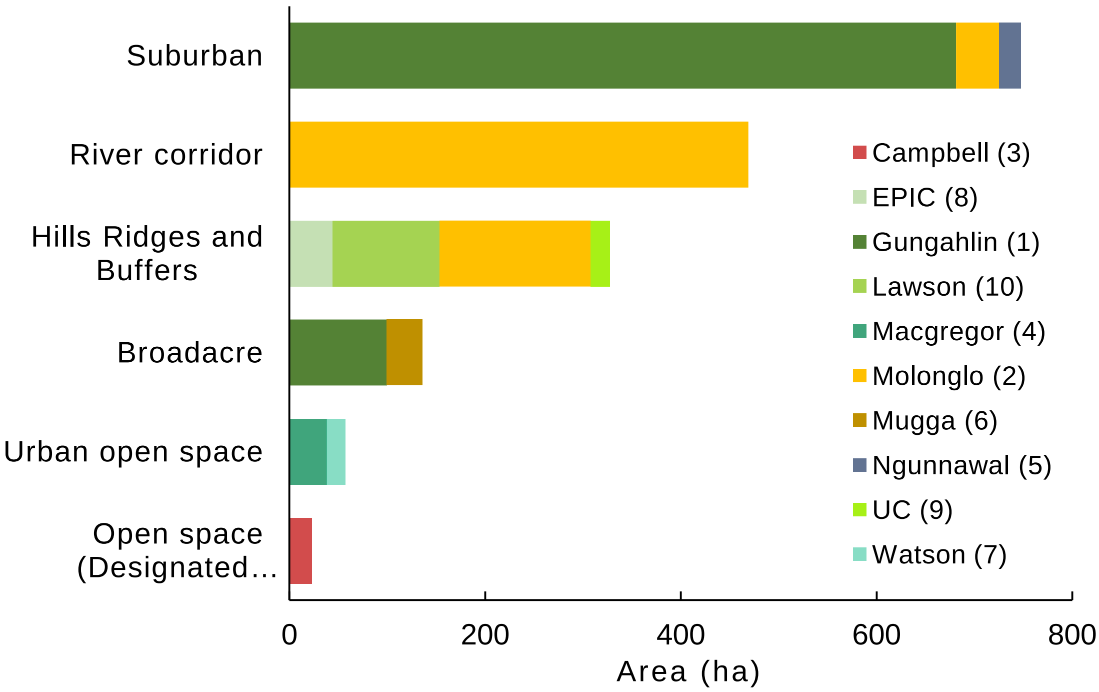


Figure 12. Amount of land (ha) associated with each land tenure type at offset sites before approval, grouped by development name and number (1-10) (see Appendix 1 for details).

Chapter 5: Discussion

In this thesis, I have provided an empirical evaluation of a biodiversity offsetting policy in order to assess the extent to which three key offsetting principles can be achieved. To do this, I evaluated offsets delivered in the ACT under the EPBC offset policy, which is the first evaluation of this policy to date. I sought to answer whether these offsets have: (1) met like-for-like requirements, (2) achieved no net loss, and (3) met the principle of additionality.

During this section, I first give a brief overview of the principles I assessed and the key conclusions from my analyses, then I interpret and discuss my findings, and conclude these sections by exploring key interpretations and highlighting recommendations supported by recent literature. Then I move on to discuss how a lack of data limited my analysis and outlining what key data and information is required for future empirical studies on offsetting.

5.1 Like-for-like

Like-for-like is a key principle utilised by offset policies and requires that offsets contain the same type of biodiversity attributes (in type, amount and condition over space and time) to those that have been lost (Maron *et al.*, 2012; Business and Biodiversity Offsets Programme, 2012). The EPBC offset policy specifically requires that offsets: (1) be in proportion to the level of statutory protection that applies to the protected matter, and (2) be of size and scale proportionate to the residual impacts on the protected matter (Australian Government, 2012a). There is evidence that at a simplistic level, the offsets examined here align with these aims. However, in relation to representations of broader vegetation types and the landscape scale context (specifically fragmentation, and implications for species richness), a successful application of the like-for-like objective is not apparent.

I found that offset ratios (the ratio of area offset to area developed) varied from 1.6:1 ha (habitat for threatened species) to 3.1:1 ha (natural temperate grassland), with a combined average of 2.3:1 ha (Table 4). These results broadly indicate that offsets are in proportion to the level of statutory protection that applies to the protected matter impacted. This means higher-level protected matters (critically endangered) have higher offset ratios than those that have a lower-level of protection (e.g. habitat for listed threatened species, where two out of the three species impacted by the developments assessed here are listed as vulnerable). However, it would be concerning if area ratios were used to conclude that good ecological outcomes are being delivered by offsets; area ratios are a harsh oversimplification of habitat and they do not give any indication of habitat quality, or the success/failure of restoration through time.

I found that development sites included a larger proportion of exotic vegetation (20%) than at offset sites (7%) (Figure 6 and Appendix 7). This generally suggests that offset sites are in better condition than development sites. Although, without comparable condition data at development sites (before impact) and offset sites, and evidence of the offset trade calculations (detailing how offsets procured gains), it is difficult to conclude whether the EPBC offset policy has promoted the simplification and substitution of site-level biodiversity attributes. For instance, whether attributes that are easier to restore place those that are more difficult (e.g. Gibbons *et al.*, 2018), or whether these calculations allowed large areas of lower biodiversity value to be commensurate with smaller areas of higher biodiversity value (e.g. Carver and Sullivan, 2017; Thorn *et al.*, 2018). I discuss the implications of limited transparency and/or lack of data further at section 5.4.

I observed that a greater amount of native grassland (non-threatened) was potentially lost at development sites (1193 ha), than gained at offset sites (689 ha) (Figure 7 and Appendix 7). This suggests that native grasslands are being heavily cleared and yet under-represented by offset requirements. It is possible that there may be inaccuracies in the spatial mapping. To rectify this, spatial mapping of the development sites and the vegetation types that occurred before any impacts would strengthen this analysis and allow for a more certain conclusion. Nevertheless, if the spatial data is accurate, it appears that much of the native grassland impacted did not require offsetting for one reason or another. One possible explanation is that the removal of these areas was not considered to be a 'significant impact' and therefore did not require offsetting (significant impact guidelines available at: Australian Government, 2013). For instance, of the 926 ha expected to be impacted due to the Gungahlin district development (total area), only 326 ha was identified as resulting in a significant impact to ecological communities or habitat for listed threatened species (Umwelt (Australia) Pty Limited, 2013). It is not clear from the documentation online why the remaining impacts were considered 'not significant', or whether these areas may, or may not provide habitat for listed species.

The three listed threatened species predominantly impacted and offset in the ACT are all grassland specialists (i.e. golden sun moth, pink-tailed worm-lizard and striped legless lizard). Albeit with different site-specific habitat requirements (i.e. the golden sun moth requires treeless open areas, the pink-tailed worm-lizard requires rocky out-crops, and striped legless lizard prefers grasslands that are structurally complex), all of these species rely heavily on a range of grassland types. In particular, recent studies on the occurrence of the golden sun moth and striped legless lizard recommend that areas of sub-optimal habitat are recognised as important for maintaining viable populations (Kutt *et al.*, 2015; Howland *et al.*, 2016). Kutt *et al.* (2015) argues that for the golden sun moth, neighbouring vegetation of identified habitat (even if sub-optimal) can act as important dispersal points for populations during wet or dry extremes. This is especially relevant considering their restricted life span (adults live for 1-4 days) and aversion to flying large distances from suitable habitat (adult males will not fly >100m) (Clarke and O'Dwyer, 2000). In addition, the golden sun moth does not exclusively occur in natural temperate grassland, and in fact uses a variety of other native grassland types (including areas dominated by Chilean needlegrass (*Nassella neesiana*) (Richter *et al.*, 2013). A recent assessment of the habitat preferences of the striped legless lizard in the ACT found that the species can persist in habitat that is floristically degraded as long as the preferred vegetation structure is present (Howland *et al.*, 2016). Howland *et al.* (2016) conclude that conservation efforts for reptile species should include smaller patches of floristically degraded habitat, not just large patches of high-quality habitat. Due to the scope of the EPBC Act, impacts to broader vegetation types and sub-optimal habitats that these studies identify as important may not necessarily be recognised or compensated for (Maron, 2014). This concern could first be addressed by the consistent reporting of all vegetation impact data at development sites, not just those deemed to be 'significant' (as I outline later in section 5.4).

At the landscape scale, my analysis suggests that development sites usually occur in more fragmented landscapes, while offsets occur in those that are more intact (Figure 8 and Table 6). This finding is consistent with that of Gibbons *et al.* (2018), who assessed offsetting outcomes in NSW that were implemented under the state's policy. It is possible that offsets are disproportionately set aside in more intact landscapes because these are areas not immediately threatened by development.

By applying the species area curve to these results, I predicted a greater impact per ha on species richness from development than gained through offsetting (Table 6 and Figure 9) (Lindenmayer and Fischer, 2006). This prediction reflects the initially steep, then plateauing trend of the species area curve (Figure 13). The curve describes how an equivalent amount of gain or loss of habitat (%) in more fragmented landscapes has a greater marginal impact on species richness overall, than in more intact landscapes (Cunningham *et al.*, 2014; Gibbons *et al.*, 2018). This indicates that that in order to fully compensate for the true loss of species richness at development sites, offsets that are set aside in more intact landscapes require a larger gain of suitable habitat (not necessarily gain in size). The additional gain required is approximately double the % of habitat lost at development sites (demonstrated in Figure 9). The feasibility of achieving this will depend on the species being impacted, expected responses to restoration measures, the characteristics of the offset site and whether restoration success is likely (Maron *et al.*, 2012). Another approach to this issue would be to prioritise sites that include less native vegetation and occur in more fragmented landscapes. While this appears counter-intuitive, in certain circumstances it could be a suitable approach to avoid overall losses. For instance, Cunningham *et al.* (2014) found that the greatest gains in bird biodiversity (per unit increase in native vegetation cover) were in areas that had low native vegetation cover. Thus, depending on the species, species richness could be safeguarded by offsetting at sites which include less native vegetation and occur in more fragmented landscapes (Cunningham *et al.*, 2014), providing that the species response to habitat creation is well-understood and the success of restoration is likely (Lindenmayer *et al.*, 2017).

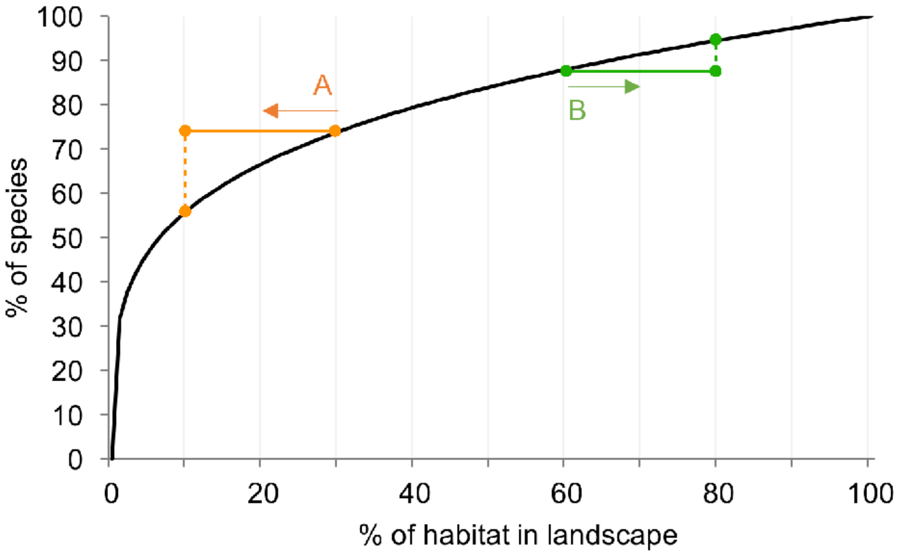


Figure 13. A generic species area curve based on the area of habitat in the landscape raised to an exponent of 0.25. The species area curve predicts that in relatively fragmented landscapes (A), the same % of habitat in the landscape lost (or gained) has a greater impact on the change in species richness than relatively intact landscapes (B). In this example, habitat is being lost in the fragmented landscape (A), and gained in the more intact (B) (denoted by the arrows), and the loss/gain for both is 20% of habitat by area.

It should be noted that the species area curve is a general representation of species-area relationships. The exact shape of the curve does differ between taxa can within taxa (Matthews *et al.*, 2014). In this case, 'habitat' is an aggregated measure that does not account for the individual habitat preferences of different species (Lindenmayer and Fischer, 2006). Moreover, the % of habitat in a landscape is only one of the factors that affects species occurrence (e.g. introduced predators, habitat quality, competition for resources) (Lindenmayer and Fischer, 2006; Cunningham *et al.*, 2014). For instance, a recent study found that the striped legless lizard is not affected by the size of habitat remnants, but rather, the density of native grazers (Howland *et al.*, 2016). Nevertheless, my analysis suggests that due to the landscape scale differences of development sites and offset sites, developments are generally resulting in greater impact per ha on species richness than gained through offset sites.

Given the implications discussed above, it is difficult to argue that offsets are delivering true like-for-like outcomes at the landscape scale. Firstly, if the spatial mapping is accurate there appears to be an overall net loss of native grassland. Even if this vegetation was sub-optimal or degraded, its loss could result in long-term impacts to grassland specialists (Tulloch *et al.*, 2016). Transparent consideration should be given to the opportunity cost associated with the destruction of sub-optimal habitat or where other habitat preferences for these animal species are present. Furthermore, this opportunity cost also needs to be considered by offset policies because the loss of a more degraded site is also the loss of potential future restoration (Peterson *et al.*, 2018). Secondly, offsetting the loss of habitat in fragmented landscapes with the protection and/or restoration in more intact landscapes appears to be leading to a net loss of species richness at the landscape scale. In order to lessen the ecological impacts of the outcomes discussed above, it is recommended that governments: (1) transparently recognise the importance of sub-optimal habitats (Tulloch *et al.*, 2016), (2) increase efforts to detect cascading effects of landscape modification before they occur (Lindenmayer and Fischer, 2006), and (3) employ more strategic approaches within jurisdictions (Gordon *et al.*, 2011).

5.2 No net loss

No net loss is an overarching objective of biodiversity offsetting, and is central to the requirement that offsets 'balance' biodiversity losses adequately (Bull *et al.*, 2013). A key characteristic of no net loss is that the 'gains' required are not intended to deliver overall conservation gains. This is because adequately implementing no net loss maintains the current trajectory (typically one of decline/ongoing loss) (Gordon *et al.*, 2015). The policy examined here aims to 'deliver an overall conservation outcome that improve or maintains the viability of the protected matter' (Australian Government, 2012a). Even though the area set aside as offsets was 2.3 times greater than the area impacted by development, it is unlikely that the policy achieved no net loss in the area of threatened vegetation for several reasons.

My analysis demonstrated that no net loss in the area of threatened vegetation (i.e. habitat for threatened species and ecological communities) cannot be met within a reasonable timeframe under most scenarios tested (Figure 10). I found that the scenarios that met no net loss within 20 years are where 50% or 75% of the offsets were based on restoration and there was a restoration success rate of ≥80% and ≥60% correspondingly. Note that: the larger proportion allocated to restoration, the smaller 'success rate' required to achieve gains within a shorter time-frame (and vice versa). If 100% of the offsets were based on restoration then they could theoretically achieve no net loss if the success of restoration within 20 years is ≥43%. However, expectations of restoration can be unrealistic and usually have a high risk of failure (Maron *et al.*, 2012), meaning that the success of restoration is generally limited (Moilanen and Kotiaho, 2018). Some studies indicate that restoration success is typically <50% (Sudol and Ambrose, 2002; Tischew *et al.*, 2010). Thus, even if 100% of the area offset was restored (1,328 ha), it is difficult to argue that this area could be successfully restored at a consistent and predictable rate that is ≥43% over 20 years.

My results reflect those Gibbons *et al.* (2018), who found that averted loss gains accrue slowly over time and risk placing the burden of current biodiversity loss onto future generations. Scenario 3 (where offsets are 25% restoration and 75% averted loss) is the closest scenario to development applications in neighbouring New South Wales (Gibbons *et al.*, 2018). Under this scenario, even with 100% restoration success, there would still be a delay of 63 years before no net loss could be met. This scenario demonstrates the long-periods over which gains associated with averted losses accrue (Figure 10). It should be noted that the analysis did not include: delays before offsets were implemented (restoration success/gain was assumed to be instantaneous at time of approval), incremental increases in gains through time, or time discounting. All of these factors are likely to have increased the time before no net loss would be achieved, and so the scenarios calculated here are expected to be best-case.

I also found that overestimating risk of loss can drastically alter when offsets appear to deliver no net loss. Maron *et al.* (2015) calculated that Australian offset policies imply crediting baselines that are on average, five times greater than recent rates of forest loss. I found that baseline stated reported by Maron *et al.* (2015) for the ACT offset policy is 125 times greater than my estimation. Assuming that 100% of the offsets are solely averting losses (no restoration), no net loss would be calculated as reached by: (a) 111 years, using the 0.004% baseline, or (b) 5 years, using the 0.5% baseline. This difference demonstrates how an ‘implausibly steep’ baseline (in this case, 0.5%) can make averted loss gains appear larger, and thus accrue within a drastically shorter timeframe (Maron *et al.*, 2015). It makes sense that a more accurate baseline would be less than estimated rates of global deforestation (0.15% per annum) (Hansen *et al.*, 2013a) or global biodiversity loss (2% per annum) (McLellan *et al.*, 2014), both of which could also be considered as implausibly steep if applied to offsetting as a generic baseline figure (Maseyk *et al.*, 2017). The perceived outcomes, and actual outcomes, of averted loss are severely impacted by the application of such baselines (Figure 14). Even if the rate applied in this case was not implausibly steep, it is fundamentally the ‘target’ implied by the policy and becomes self-reinforcing in nature (or ‘locked in’) (Gordon *et al.*, 2015), whether realistic or not (Maron, 2014). Thus, my findings endorse earlier warnings that gains from averted loss can be easily overstated, which is a key flaw of averted loss offsets (Gordon *et al.*, 2015; Maron *et al.*, 2015; Gibbons *et al.*, 2016).

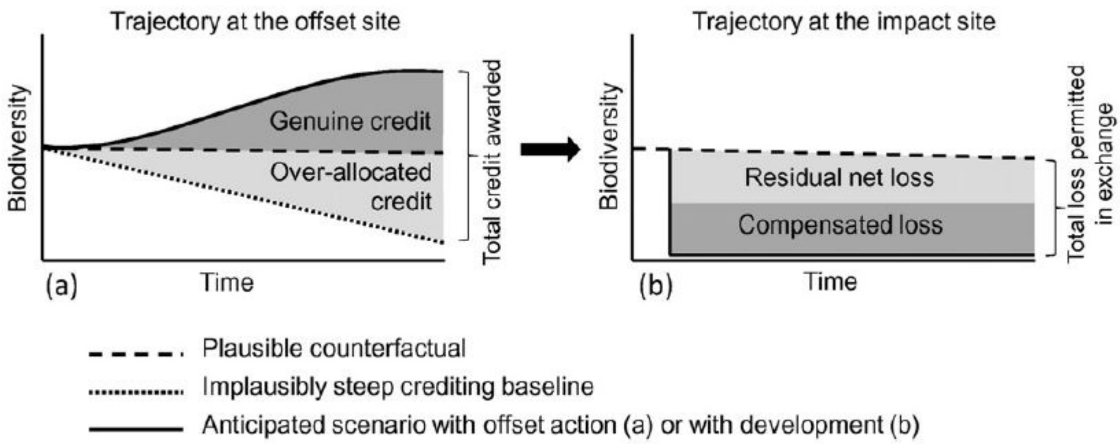


Figure 14. Example of offset calculation/trade where an implausibly steep crediting baseline is used (fine dotted line), which results in: (a) an increase of offset credit, and (b) an exchange for equivalent loss. In (a), the dark grey represents genuine credit (plausible counterfactual), and over-allocated credit (light grey), resulting in a corresponding, residual net loss due to over allocation in (b) (light grey shading) (Maron *et al.*, 2015).

A key consideration is that while proponents must explicitly 'risk of loss' estimates in order to calculate offset requirements, these are not publically available and recent advice indicates that they are being calculated inconsistently due to inexplicit guidance (Maseyk *et al.*, 2017). Given that the EPBC offset policy incorrectly recommends the inclusion of impacts already covered by the policy in risk of loss estimates (i.e. site counterfactuals), it is likely that in reality, estimates used are significantly higher (or 'steeper') than the estimate I used for the analysis (0.004%). The guidance states that risk of loss estimates can include 'pending development applications, mining leases or other activities on the proposed offset site that indicate development intent and likelihood' (Australian Government, 2012b). This is a prominent issue in offset policies (Peterson *et al.*, 2018; Maron *et al.*, 2018; Maron, 2014; Maseyk *et al.*, 2017). By incorporating impacts already covered by the policy, the overall offset requirement is decreased due to false gains (see example in detail at Appendix 11). This is because gains from the protection of a site (averted loss) are not applicable where impacts that are covered by the policy threatened the site and will trigger an offset (i.e. urban development) (Maseyk *et al.*, 2017). For example, if urban development did end up occurring at a proposed offset site which contained habitat for threatened species, then that loss would require an offset under the policy (Maron *et al.*, 2018; Peterson *et al.*, 2018; Maseyk *et al.*, 2017). Effectively, there is no averted loss from the protection of the proposed offset site. No net loss calculations (or what is deemed 'no net loss') are greatly influenced by the calculation of site counterfactuals (i.e. risk of loss) and reference scenarios. (Gordon *et al.*, 2011; Peterson *et al.*, 2018). Since governments are tasked with either their calculation or validation, it is essential that the application of these scenarios, methods advised for their calculation, and the figures themselves, are both explicit and available for public scrutiny (Maron *et al.*, 2018; Maron *et al.*, 2016a; Gordon *et al.*, 2015).

Perhaps *the* most vital concern regarding declining reference scenarios is their incredibly broad application in the absence of sufficient justification. I argue that it is nonsensical for an impact-specific, no net loss policy to assume any rate of decline (or crediting baseline) under the reference scenario, which partially opposes arguments made by Peterson *et al.* (2018). I agree that site counterfactuals (impact-specific) are only applicable where impacts that are not covered by the offsetting policy or other legislation threaten the site (discussed above) (Maron *et al.*, 2018; Maron *et al.*, 2015). However, in regard to the inclusion of 'background' biodiversity decline, or 'declining baselines', I propose that these estimates, more often than not, should not be included in averted loss calculations. In particular, Maron *et al.* (2015) rationalise that a baseline of decline is only applicable if the decline is expected regardless of development impacts: for example, if the decline in population of a particular species can be attributed to an introduced predator. While this is logical, Maron *et al.* (2015) and Peterson *et al.* (2018) fail to discuss the difficulties in isolating 'background' biodiversity decline, or the 'background' decline of a species (where 'background' means threats not associated with impacts from development, such as invasive species or climate change). The only apparent benefit of applying background biodiversity decline, or declining baselines, then, would appear to be in order to reduce and relax offset requirements so that averted loss offsets deliver superficial gains. This would allow ongoing impacts from development, while offset policies appear to be achieving 'no net loss' when in fact they support the trajectory of ongoing biodiversity decline (Maseyk *et al.*, 2017). Thus, I argue that the use of a declining baseline (or applying a high 'risk of loss') is inappropriate unless there is substantial evidence that the estimated declining trend is in no way connected to the impacts covered by the offsetting policy. Following this principle, for both reference scenarios (i.e. baselines) and site-level counterfactuals, it is likely that averted loss gains are not actually resulting in true gains under most circumstances, or at least the majority of which averted loss is applied (Maron *et al.*, 2012).

If it is recognised that if there is no loss under a realistic counterfactual, then no offset credit can be obtained through averting loss, and thus, achieving no net loss depends entirely on the success of restoration. Adequate implementation, or the 'effectiveness' (not just 'compliance') of restoration faces multiple challenges (Lindenmayer *et al.*, 2017; May *et al.*, 2017; Sudol and Ambrose, 2002). May *et al.* (2017) found that past restoration offsets had the highest proportion of 'no outcome', 'too early to tell' and 'unknown' outcomes. Another study that evaluated the success of a 'well-resourced and well-documented' restoration project found that there was either a complete failure of the restoration actions, or the restoration trajectory is so slow to detect a change within 10 years of establishment (Wilkins *et al.*, 2003). To minimise risks of restoration failure, more long-term offset monitoring projects are required, and these will need to include ways of testing the success of restoration measures through time (Lindenmayer *et al.*, 2017; Ferraro and Pattanayak, 2006; Maron *et al.*, 2012).

Therefore, if 'averted loss' is an illegitimate way to accrue offset gains, and restoration is peppered with major risks, uncertainty and time lags (Maron *et al.*, 2012), adequately balancing losses with gains in order to achieve 'no net loss' is a highly challenging aim (Wilkins *et al.*, 2003; Thorn *et al.*, 2018).

5.3 Additionality

In the EPBC offset policy additionality it is defined as 'additional to what is already required, determined by law or planning regulations or agreed to under other schemes or programs' (Australian Government, 2012a). From my results, I could not determine with confidence that offsets in the ACT have achieved, or failed to meet the requirements of additionality. Nonetheless, two main concerns arose, specifically: (1) the EPBC offset policy incorrectly frames the principle of additionality, and (2) there is a lack of information on existing commitments within various land-use zones and protected areas.

I found that while there is no firm indication that offsets have displaced the frequency of reservation, there has been a decline in the amount of area reserved in recent years (Figure 11). Five years before biodiversity offsetting was introduced (2005-2009), 1,495 ha was set aside for reservation, while five years after (2010-2014), a total of 446 ha was reserved. The latter figure excludes the addition of 308 ha allocated to the reserve system that are offsets. While there is a large difference between these figures, additional area incorporated into the reserve system each year is generally sporadic (Appendix 9). It is therefore difficult to argue based on these findings that the introduction of offsetting has directly caused a declining trend in the land allocated to reserves.

It was not possible to determine whether there is a trend through time in in the change of funding (allocated to the Parks and Conservation Service for managing offsets). While these data were summarised (Appendix 10), I was unable to isolate a financial trend in relation to offsets. There are two key reasons why: (1) due to the changing responsibilities and structure of the Directorate through time, and (2) the lack of specificity of budget allocation in relation to managing reserves and offsets. In addition, financial data were only available for several of the offsets on the Offset Register (Appendix 10). Thus, I was unable to determine whether offsets have displaced the reserve program in the ACT, or caused a winding back of other conservation actions (i.e. resulted in 'cost shifting') (Gordon *et al.*, 2015; Pilgrim and Bennun, 2014). Still, this should not deter from the fact that there are incentives for governments to cost-shift where funding is cut or limited (Githiru *et al.*, 2015), along with existing pressure to meet targets (Narain and Maron, 2018).

My analysis of offset sites by land tenure showed that a large proportion (42%) of the overall area offset in the ACT were zoned as suburban areas before approval (Table 7). This statistic is almost completely in relation to offsets associated with the Gungahlin development (Figure 12). This result encourages consideration of whether averted loss gains were calculated on the basis that these areas were planned for future suburban development. Currently, the EPBC offset policy partly frames additionality as concerning whether or not the site 'can be built upon due to zoning laws' (i.e. '*is future development possible or likely?'*) (Australian Government, 2012a). Framing additionality in this way is misleading because site counterfactuals should not include actions which themselves would trigger offset requirements (Maron *et al.*, 2016a) (as discussed in section 5.2). Fulfilling additionality then relates solely to whether the site is being managed, or intends to be for existing conservation commitments not related to offsetting (i.e. the management of protected areas such as nature reserves and national parks) (Maseyk et al., 2017).

My analysis of offsets by land tenure also showed that river corridors (27%) were the second highest zone offset (which was largely associated with offsets related to the Molonglo development), followed by hills ridges and buffers (19%) (Table 7 and Figure 12). Both of these non-urban zones have general disturbance restrictions that intend 'to protect woodlands, native grasslands, forests and waterways' (ACT Planning & Land Authority, 2013). Specifically, the hills, ridges and buffer areas are planned to 'conserve the significant cultural and natural heritage resources and a diversity of habitats and wildlife corridors' (this zone includes all Nature Reserves), and river corridors have restricted to use to 'ensure development is kept to a minimum... and is confined to the perimeter of environmentally sensitive areas' (ACT Planning & Land Authority, 2013). So, while there is a general conservation focus for these two zones, there is a lack of specificity regarding what levels of management are implemented or required (Gibbons, 2011). Without knowledge of existing commitments within these zones, it is difficult to determine whether offset actions in these areas are 'above and beyond' the duty of care imposed by state-legislation (Gibbons, 2011). From the documentation available, I was unable to find evidence of any offset management plans that recognise the current level of management in these zones, or how the offsets are additional to existing requirements.

Similarly, I also observed that from the documentation available that it was difficult to determine how actions in protected areas (for instance, Kama Nature Reserve) deviated 'significantly' from those already required or expected. In particular, the creation of a management plan targeted specifically to the offsets in the Molonglo region (including Kama Nature Reserve) was proposed as part of the offset package. The intention of these plans were to 'ensure consistency of objectives and management for each of the 'matters of national environmental significance' while reducing the risk of fragmentation and inconsistent management approaches for the difference locations' (ACT Government, 2013). Without evidence of any consideration of existing management commitments, or an explicit statement that these plans were unlikely to be created in absence of the offset, it is difficult to conclude that adequate due diligence has been taken in order to meet additionality.

Generally, authors agree that the use of biodiversity offsetting to fund protected, or semi-protected areas is justifiable as long as the financing/restoration efforts proposed are unlikely to be allocated otherwise (Pilgrim and Bennun, 2014; Githiru *et al.*, 2015; Maron *et al.*, 2016a). Maron *et al.* (2016a) provide a decision tree that demonstrates the circumstances in which these actions would still be considered 'valid' in meeting additionality (Figure 15). Seeing as financial deficits in the funding of protected areas is most apparent in developing countries (Githiru *et al.*, 2015), it seems appropriate that governments within developed countries incur greater scrutiny when proposing to undertake management actions in already protected areas, and should provide sufficient evidence that existing, or potential management actions/funding, is not replaced the offset funding (Pilgrim and Bennun, 2014; Githiru *et al.*, 2015; Maron *et al.*, 2016a). Although, in order for this to occur, governments require some 'political will' to reveal management and financial plans, provide transparent analyses of projected costs and outputs (Githiru *et al.*, 2015), and allow public scrutiny of these plans and their implementation (Pilgrim and Bennun, 2014).

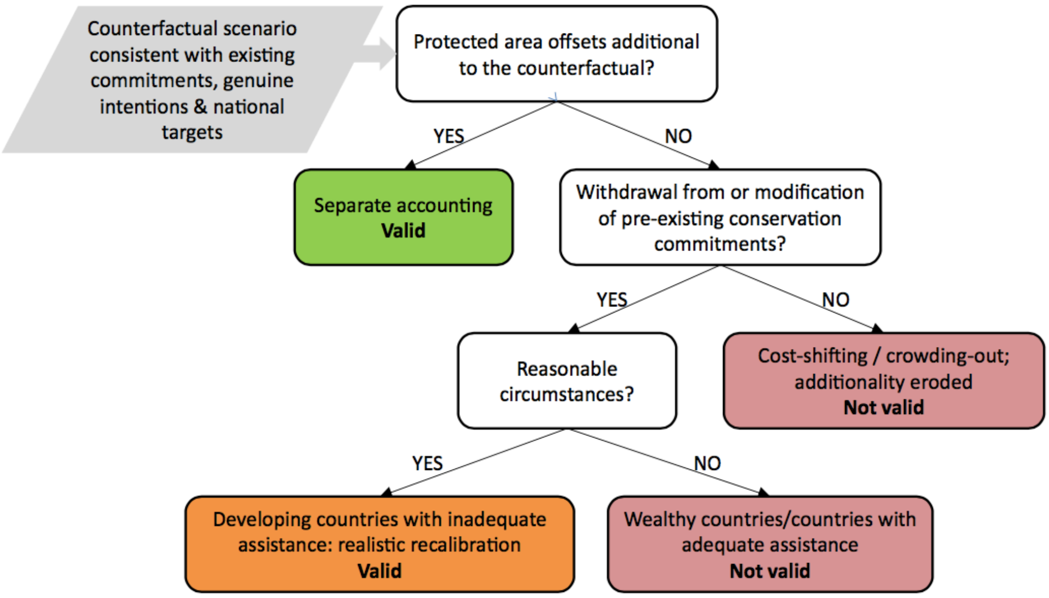


Figure 15. Decision tree indicating the validity of offset funding being used to fund the purchase and/or management of protected areas, with distinction of different appropriateness depending a countries capacity/ability to meet commitments (Maron *et al.*, 2016a). Where 'counterfactual scenario' is a reflection of what would have occurred at the site level.

While I was unable to determine whether offsets in the ACT meet the principle of additionality, two key concerns arose from my analyses. Firstly, additionality is framed incorrectly in the EPBC offset policy. The policy recommends considering whether there is risk of development at offset sites without specifying that this is only applicable when the potential development would not trigger an offset (which is rare) (Maseyk *et al.*, 2017). Secondly, where offsets have occurred in zones that are already protected and/or managed by the governments for conservation purposes, there is a lack of explicit recognition regarding how these offset actions are 'above and beyond' what is currently required. It is likely that these issues have arisen since there is a lack of incentive for governments to disclose the site counterfactuals used for offsets, or openly report the financial plans of existing conservation management (Githiru *et al.*, 2015; Pilgrim and Bennun, 2014). Ultimately, these issues could be resolved based on recommendations made by the literature to date (Gordon *et al.*, 2015; Narain and Maron, 2018; Githiru *et al.*, 2015; Pilgrim and Bennun, 2014; Maron *et al.*, 2016a). In particular, adoption of current guidance (e.g. Maseyk *et al.*, 2017); Maron *et al.* (2016a) and transparent accounting (i.e. offset calculations that are publically available which clearly distinguish site counterfactuals and assumptions made).

5.4 Data availability

The single biggest limitation of my research, and fundamental topic for consideration, is the lack of data available on the developments and offsets examined here. The insufficient collection and/or reporting of data is a mutual concern for many authors when reviewing the outcomes of individual projects (Lindenmayer *et al.*, 2017) and offset policies (May *et al.*, 2017; Gibbons *et al.*, 2018; Bezombes *et al.*, 2019; Bull *et al.*, 2018). All generally recommend a more systematic approach to the collection of data, better organisation and greater accessibility.

It seems that the standard for reporting key offset data is generally low. A recent study of offsets in France found that of the 91 offset procedure files examined, only 33% contained sufficient information to be included in the study, and all of these did not include details on the initial state of the offset site (Bezombes *et al.*, 2019). May *et al.* (2017) were unable to assess whether offsets in Western Australia effectively counterbalanced impacts, due to missing impact area data. Gibbons *et al.* (2018) had sufficient data to undertake their offset analysis, pointed to a lack of information on the change in habitat condition resulting from biodiversity offsets.

In relation to the offsets reviewed here, the most recent ACT State of the Environment Report recognises that 'insufficient data have been collected to assess the cumulative impacts of multiple offsets' (Office of the Commissioner for Sustainability and the Environment, 2015). A stakeholder analysis based on the EPBC offset policy inquiry established that 'the systematic failure of monitoring and compliance activity is arguably a contributing factor in a general absence of a comprehensive, scientific dataset on the impact of offsetting on environmental management' (Martin *et al.*, 2016).

I found that while some documents are available on the offset register (i.e. offset management plans, compliance reports, monitoring reports), these are not provided consistently for all offsets. Perhaps more importantly, there is no explanation as to why this is the case. I found that while some more comprehensive monitoring reports have been made available online for specific matters (SMEC Australia, 2017; SMEC Australia, 2015), these do not appear to have been replicated through time, although there seems to be future intentions to do so.

The following are the key data that would add significant value to my research (and, any review of biodiversity offsets):

a) spatial mapping of development site before impact and after, including all impacts to vegetation (not just those deemed 'significant')
b) condition data (preferably in spatial mapping form) for impact and offset sites prior to development/offsetting
c) offset trade calculations (i.e. EPBC 'guide' inputs if used for calculations, and documentation of how additional measures, indirect offsets, or other negotiations that altered offset requirements, e.g. Carver and Sullivan (2017)), with a clear distinction of what proportion of offset gains are associated with proposed restorative measures and what level of overall 'success' is required, as well as:
    - counterfactuals and/or crediting baselines applied
    - explicit statement of assumptions and any evidence to validate estimations
d) site monitoring reports undertaken periodically and with consideration of landscape scale changes, weather, and restoration measures (if applicable) applied

With these data, one could undertake an empirical evaluation of offsets delivered by a policy, or individual offset project, and be able to determine whether: (1) the offset requirements have been calculated 'correctly' (in accordance to the policy), and (2) there are potential gaps or net losses.

For a more comprehensive understanding of offsetting outcomes (specifically in relation to the principles assessed here), data are also required beyond the impact/offset sites themselves. In particular, I identified that explicit information on the broader conservation commitments/intentions of the government (e.g. management activities and financing) was missing or unclear. As recognised by Ferraro and Pattanayak (2006), properly evaluating conservation interventions requires a combination of data, including 'ecological, geographic, socio-economic, demographic, and institutional measures'. Adequate data collection and monitoring will only be achieved if it is a planned component of offsetting policies (Ferraro and Pattanayak, 2006) and is undertaken routinely with the purpose of evaluation in mind (Dovers and Hussey, 2013).

Chapter 6: Conclusion

Urgent efforts to slow key drivers of biodiversity loss are vital (Díaz *et al.*, 2019). Of particular concern is the loss and fragmentation of habitat caused by land clearing (Hansen *et al.*, 2013a; Evans, 2016). Biodiversity offsetting is a relatively new, yet widely adopted policy mechanism that attempts to balance the ongoing loss of habitat by requiring the protection and sometimes rehabilitation of habitat elsewhere. To date, over 80 countries have biodiversity offsetting policies in place (Maron *et al.*, 2018). Empirical evaluations of these policies are lacking and therefore, the extent to which the key principles of offsetting are being implemented is not well known. By looking at offsets delivered under the Australia's federal offset policy in the ACT, I evaluated the extent to which these offsets: (1) are equivalent or like-for-like, (2) have achieved no net loss and (3) are additional. Overall, I found there are multiple issues associated with the implementation and evaluation of these principles.

I found that meeting like-for-like under the EPBC offset policy may not adequately account for the importance of 'sub-optimal' habitat for species (in this case: non-threatened habitat for grassland specialists), which conflicts with the policy's aim that offsets 'be of size and scale proportionate to the residual impacts on the protected matter' (Australian Government, 2012a). Since the offsets reviewed here are not achieving like-for-like at the landscape scale, I also found that offsets may exacerbate loss within areas that are already heavily cleared (Gibbons *et al.*, 2018).

No net loss is an aim yet to be demonstrated possible in scenario modelling (Maron *et al.*, 2010; Sonter *et al.*, 2017), or in practice (Quigley and Harper, 2006; May *et al.*, 2017; Gibbons *et al.*, 2018; Bezombes *et al.*, 2019; Thorn *et al.*, 2018). I found that the offsets reviewed here are unlikely to meet no net loss within a suitable timeframe unless the majority of offsets are based on restoration, and the success of restoration is unrealistically high. The inability to achieve no net loss (in area of vegetation) can be attributed to two key challenges faced when calculating the gains of offsets: (1) the correct application of averted losses (particularly the calculation of baselines and site counterfactuals) and (2) assumptions on the success of restoration. I argued that the use of declining baselines is inappropriate unless there is substantial evidence that the estimated declining trend of the threatened vegetation is not connected to the impacts covered by the offset policy (Maron *et al.*, 2018) (this also relates to my findings on additionality).

While I could not determine whether offsets in the ACT have achieved additionality, I identified two main concerns related to implementing and evaluating this principle. First, the EPBC offset policy frames the additionality by asking the question 'is future development possible or likely?' (Australian Government, 2012a). This incorrectly encourages the calculation of gains from averted loss where no gains are actually occurring. Second, there is inadequate transparency regarding counterfactual scenarios. Particularly, the existing financial and management commitments within various land-use zones and protected areas that offsets are applied.

It became apparent during this evaluation that the three key principles I assessed do not act in isolation. The fulfilment of one principle will have flow-on effects to another. For instance, achieving additionality will directly relate to meeting no net loss. As empirical evaluations of offset policies improve and the policies themselves are revised, it is expected that the connections between these principles will be made clear and potentially merged into new definitions where appropriate (so as to limit confusion).

The single biggest limitation of this study was the lack of data available. The main data missing from this analysis was the offset trade calculations, particularly the amount of gains accrued from averted loss versus restoration and the baselines and/or site counterfactuals used. Still, there are two key benefits from undertaking this evaluation using limited data. Firstly, the methods used can be easily applied to other jurisdictions where data is also incomplete or missing. Secondly, I was able to identify specifically what data would support future evaluations (outlined in section 5.4). In alignment existing recommendations, I concluded that data collection needs to be strategic, routine and accessible.

I recognise that there are multiple other theoretical and practical challenges faced by biodiversity offsetting that I have not discussed due to the limited scope of this thesis (Maron *et al.*, 2016b). In particular: metric choice and trade negotiations (Carver and Sullivan, 2017), offset longevity (Bull *et al.*, 2013), costs (Lindenmayer *et al.*, 2017), the application of indirect offsets (i.e. financing research), consistency between offsets and national conservation targets/goals (Maron *et al.*, 2018), and governance (Maron *et al.*, 2016b). I acknowledge that these are critical topics of focus for future studies on offset policies, including the EPBC offset policy. I also recognise the importance of qualitative evaluations in providing valuable understanding of 'why' offset policies are, or are not working (Evans, 2017).

Assessing the effectiveness of biodiversity offsetting policies remains profoundly complicated. Robust evaluations of offsetting policies are only possible if sufficient data is collected and managed correctly. Without these efforts, detecting the overall outcomes of offsetting policies through time, and thus determining whether they are better than no intervention at all, will remain difficult. The emerging site level (Lindenmayer *et al.*, 2017; Thorn *et al.*, 2018) and policy level evaluations (Gibbons *et al.*, 2018; Bezombes *et al.*, 2019; May *et al.*, 2017), including this thesis, demonstrate that there are significant challenges in achieving the fundamental principles of biodiversity offsetting. Thus, in order to support the continuing use of biodiversity offsetting as a mechanism to 'balance' irreversible impacts on biodiversity, we need more evidence and greater certainty that these policies are actually working. By reviewing offsets delivered in the ACT under Australia's federal offset policy, I have supported, and expanded on, the present challenges faced when implementing and evaluating biodiversity offset policies.

References

ACT Government, 2013. *Molonglo Adaptive Management Strategy*, Available at: http://www.tams.act.gov.au/__data/assets/pdf_file/0003/588045/Molonglo-Adaptive-Management-Strategy.pdf

ACT Government, 2014. *ACT Offsets Policy and Delivery Framework - Position Paper*, Available at: https://www.environment.act.gov.au/cpr/environmental-offsets-policy/act-offsets-policy-and-delivery-framework-position-paper

ACT Planning & Land Authority, 2013. (ed, ACT Parliamentary Counsel).

Apostolopoulou, E. and Adams, W. M., 2017. Biodiversity offsetting and conservation: reframing nature to save it, *Oryx*, **51**(1): 23-31. Available at: 10.1017/S0030605315000782

Australian Bureau of Agricultural and Resource Economics and Sciences, 2018a. *Australian land use profiles 2017*, Available at: https://public.tableau.com/profile/australian.bureau.of.agricultural.and.resource.economics.and.sci - !/vizhome/Landuseprofiles2017v2/Story

Resources, D. o. A. a. W., *Snapshot of Australian Agriculture*,

Australian Bureau of Statistics, 2019. *Regional Population Growth, Australia, 2017-18*, Available at: http://www.abs.gov.au/ausstats/abs@.nsf/mf/3218.0 (accessed 04/04/19).

Australian Government, *Strategic assessments* Available at: https://environment.gov.au/protection/assessments/strategic

Resources, D. o. E. a. W., *Draft Policy Statement: Use of environmental offsets under the Environment Protection and Biodiversity Conservation Act 1999*,

Department of Sustainability, E., Water, Population and Communities,, *Environment Protection and Biodiversity Conservation Act 1999 Environmental Offsets Policy*, Commonwealth of Australia,,

Australian Government, 2012b. EPBC offset policy: how to use the offsets assessment guide. Available at: http://www.environment.gov.au/system/files/resources/12630bb4-2c10-4c8e-815f-2d7862bf87e7/files/offsets-how-use.pdf

Department of the Environment, *Matters of National Environmental Significance: Significant impact guidelines 1.1*,

Australian Government, 2018a. *About the EPBC Act*, Available at: http://www.environment.gov.au/epbc/about

Department of Environment and Energy, *Annual Report 2017-18 - Appendices*, Commonwealth of Australia,

Australian State of the Environment Committee, 2001. *Australia State of the Environment 2001: Independent Report to the Commonwealth Minister for the Environment and Heritage*, Publishing on behalf of the Department of the Environment and Heritage, Canberra, CSIRO.

Bezombes, L., Kerbiriou, C. and Spiegelberger, T., 2019. Do biodiversity offsets achieve No Net Loss? An evaluation of offsets in a French department, *Biological Conservation*, **231**: 24-29. Available at: 10.1016/j.biocon.2019.01.004

Bonneuil, C., 2015. Tell me where you come from, I will tell you who you are: A genealogy of biodiversity offsetting mechanisms in historical context, *Biological Conservation*, **192**: 485-491. Available at: 10.1016/j.biocon.2015.09.022

Bull, J. W., Brauneder, K., Darbi, M., Van Teeffelen, A. J. A., Quétier, F., Brooks, S. E., Dunnett, S. and Strange, N., 2018. Data transparency regarding the implementation of European 'no net loss' biodiversity policies, *Biological Conservation*, **218**: 64-72. Available at: 10.1016/j.biocon.2017.12.002

Bull, J. W., Milner-Gulland, E. J., Suttle, K. B., Singh, N. J. and Sveriges, l., 2014. Comparing biodiversity offset calculation methods with a case study in Uzbekistan, *Biological Conservation*, **178**: 2-10. Available at: 10.1016/j.biocon.2014.07.006

Bull, J. W., Suttle, K. B., Gordon, A., Singh, N. J. and Milner-Gulland, E. J., 2013. Biodiversity offsets in theory and practice, *Oryx*, **47**(3): 369-380. Available at: 10.1017/S003060531200172X

Business and Biodiversity Offsets Programme, 2012. *Guidance notes to the Standard on Biodviersity Offsets*, Washington, D.C. Available at: https://www.forest-trends.org/wp-content/uploads/imported/BBOP_Standard_Guidance_Notes_20_Mar_2012_Final_WEB.pdf

Business and Biodiversity Offsets Programme, 2013. *To no net loss and beyond: an overview of the Business and Biodiversity Offsets Programme (BBOP)*, Washington, D.C. Available at: https://www.forest-trends.org/wp-content/uploads/imported/bbop-overview-document_2012_v11_april-22_2013_web-pdf.pdf

Butchart, S. H. M., Walpole, M., Collen, B., Strien, A. v., Jörn, P. W. S., Rosamunde, E. A. A., Jonathan, E. M. B., Bomhard, B., Brown, C., Bruno, J., Carpenter, K. E., Carr, G. M., Chanson, J., Chenery, A. M., Csirke, J., Davidson, N., Dentener, F., Foster, M., Galli, A., Galloway, J. N., Genovesi, P., Gregory, R. D., Hockings, M., Kapos, V., Lamarque, J.-F., Leverington, F., Loh, J., McGeoch, M. A., McRae, L., Minasyan, A., Morcillo, M. H., Thomasina, E. E. O., Pauly, D., Quader, S., Revenga, C., Sauer, J. R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S. N., Symes, A., Tierney, M., Tyrrell, T. D., Vié, J.-C. and Watson, R., 2010. Global Biodiversity: Indicators of Recent Declines, *Science*, **328**(5982): 1164-1168. Available at: 10.1126/science.1187512

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., Narwani, A., Mace, G. M., Tilman, D., Wardle, D. A., Kinzig, A. P., Daily, G. C., Loreau, M., Grace, J. B., Larigauderie, A., Srivastava, D. S. and Naeem, S., 2012. Biodiversity loss and its impact on humanity, *Nature*, **486**(7401): 59-67.

Carver, L. and Sullivan, S., 2017. How economic contexts shape calculations of yield in biodiversity offsetting, *Conservation Biology*, **31**(5): 1053-1065. Available at: doi:10.1111/cobi.12917

Chapin, F. S., III, Zavaleta, E. S., Eviner, V. T., Naylor, R. L. and et al., 2000. Consequences of changing biodiversity, *Nature*, **405**(6783): 234-242. Available at: http://dx.doi.org/10.1038/35012241

Clarke, G. M. and O'Dwyer, C., 2000. Genetic variability and population structure of the endangered golden sun moth, Synemon plana, *Biological Conservation*, **92**(3): 371-381. Available at: 10.1016/S0006-3207(99)00110-X

Commonwealth of Australia, 2014. *Senate enquiry into environmental offsets*, Canberra. Available at: https://www.aph.gov.au/Parliamentary_Business/Committees/Senate/Environment_and_Communications/Environmental_Offsets/Report/~/media/Committees/Senate/committee/ec_ctte/environ_offset/report/report.pdf

Connor, E. F. and McCoy, E. D., 1979. The Statistics and Biology of the Species-Area Relationship, *The American Naturalist*, **113**(6): 791-833. Available at: 10.1086/283438

Cresswell, I. D. and Murphy, H. T., 2017. *Australia state of environment 2016: biodiversity, independent report to the Australian Government Minister for the Environment and Energy*, Canberra.

Cunningham, R. B., Lindenmayer, D. B., Crane, M., Michael, D. R., Barton, P. S., Gibbons, P., Okada, S., Ikin, K. and John, A. R. S., 2014. The law of diminishing returns: woodland birds respond to native vegetation cover at multiple spatial scales and over time, *Diversity and Distributions*, **20**(1/2): 59-71. Available at: 10.1111/ddi.12145

Díaz, S., Settele, J. and Brondízio, E., 2019. *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*, Available at: https://www.ipbes.net/sites/default/files/downloads/spm_unedited_advance_for_posting_htn.pdf

Dovers, S. and Hussey, K., 2013. *Environment and sustainability: a policy handbook*, The Federation Press, Sydney, Australia.

Environment ACT, 2005. *National Recovery Plan for Natural Temperate Grassland of the Southern Tablelands (NSW and ACT): an endangered ecological community*, Environment ACT, Canberra.

Evans, M. C., 2016. Deforestation in Australia: drivers, trends and policy responses, *Pacific Conservation Biology*, **22**(2): 130-150. Available at: https://doi.org/10.1071/PC15052

Evans, M. C., 2017. 'Public policy for biodiversity conservation: evaluating outcomes, opportunities and risks', The Australian National University,

Ferraro, P. J. and Pattanayak, S. K., 2006. Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments, *PLOS Biology*, **4**(4): e105. Available at: 10.1371/journal.pbio.0040105

Fischer, J. and Lindenmayer, D. B., 2007. Landscape Modification and Habitat Fragmentation: A Synthesis, *Global Ecology and Biogeography*, **16**(3): 265-280. Available at: 10.1111/j.1466-8238.2007.00287.x

Food and Agriculture Organization of the United Nations, 2018. The future of food and agriculture: Alternative pathways to 2050.

Gibbons, P., 2011. *Potential biodiversity offset actions and sites for the Australian Capital Territory*, Available at: http://www.environmentcommissioner.act.gov.au/__data/assets/pdf_file/0008/590975/ocse_potential_biodiversity_offset_mgt_march_2011.pdf

Gibbons, P., Evans, M. C., Maron, M., Gordon, A., Roux, D., Hase, A., Lindenmayer, D. B. and Possingham, H. P., 2016. A Loss-Gain Calculator for Biodiversity Offsets and the Circumstances in Which No Net Loss Is Feasible, *Conservation Letters*, **9**(4): 252-259. Available at: doi:10.1111/conl.12206

Gibbons, P. and Lindenmayer, D., 2007. Offsets for land clearing: No net loss or the tail wagging the dog?, *Ecological Management & Restoration*, **8**(1): 26-31. Available at: 10.1111/j.1442-8903.2007.00328.x (accessed 2018/07/27).

Gibbons, P., Macintosh, A., Constable, A. L. and Hayashi, K., 2018. Outcomes from 10 years of biodiversity offsetting, *Global Change Biology*, **24**(2): e643-e654. Available at: doi:10.1111/gcb.13977

Githiru, M., King, M. W., Bauche, P., Simon, C., Boles, J., Rindt, C. and Victurine, R., 2015. Should biodiversity offsets help finance underfunded Protected Areas?, *Biological Conservation*, **191**: 819-826. Available at: https://doi.org/10.1016/j.biocon.2015.07.033

Gordon, A., Bull Joseph, W., Wilcox, C. and Maron, M., 2015. FORUM: Perverse incentives risk undermining biodiversity offset policies, *Journal of Applied Ecology*, **52**(2): 532-537. Available at: 10.1111/1365-2664.12398 (accessed 2018/07/22).

Gordon, A., Langford, W. T., Todd, J. A., White, M. D., Mullerworth, D. W. and Bekessy, S. A., 2011. Assessing the impacts of biodiversity offset policies, *Environmental Modelling and Software*, **26**(12): 1481-1488. Available at: 10.1016/j.envsoft.2011.07.021

Territory Plan,

Guillet, F. and Semal, L., 2018. Policy flaws of biodiversity offsetting as a conservation strategy, *Biological Conservation*, **221**: 86-90. Available at: 10.1016/j.biocon.2018.03.001

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S., Tyukavina, A., Thau, D., Stehman, S., Goetz, S. and Loveland, T. R., 2013a. High-resolution global maps of 21st-century forest cover change, *Science*, **342**(6160): 850-853.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O. and Townshend, J. R. G., 2013b. High-Resolution Global Maps of 21st-Century Forest Cover Change, *Science*, **342**(6160): 850-853. Available at: 10.1126/science.1244693

Hoffmann, M., Hilton-Taylor, C., Angulo, A., Böhm, M., Brooks, T. M., Stuart, H. M. B., Carpenter, K. E., Chanson, J., Collen, B., Cox, N. A., William, R. T. D., Dulvy, N. K., Harrison, L. R., Katariya, V., Pollock, C. M., Quader, S., Richman, N. I., Ana, S. L. R., Tognelli, M. F., Vie, J.-C., Aguiar, J. M., Allen, D. J., Allen, G. R., Amori, G., Ananjeva, N. B., Andreone, F., Andrew, P., Aida Luz Aquino, O., Jonathan, E. M. B., Baldi, R., Bell, B. D., Biju, S. D., Bird, J. P., Black-Decima, P., Blanc, J. J., Bolaños, F., Bolivar-G, W., Burfield, I. J., Burton, J. A., Capper, D. R., Castro, F., Catullo, G., Cavanagh, R. D., Channing, A., Chao, N. L., Chenery, A. M., Chiozza, F., Clausnitzer, V., Collar, N. J., Collett, L. C., Collette, B. B., Claudia, F. C. F., Craig, M. T., Crosby, M. J., Cumberlidge, N., Cuttelod, A., Derocher, A. E., Diesmos, A. C., Donaldson, J. S., Duckworth, J. W., Dutson, G., Dutta, S. K., Emslie, R. H., Farjon, A., Fowler, S., Freyhof, J., Garshelis, D. L., Gerlach, J., Gower, D. J., Grant, T. D., Hammerson, G. A., Harris, R.

B., Heaney, L. R., Hedges, S. B., Hero, J.-M., Hughes, B., Hussain, S. A., M, J. I., Inger, R. F., Ishii, N., Iskandar, D. T., Richard, K. B. J., Kaneko, Y., Kottelat, M., Kovacs, K. M., Kuzmin, S. L., Marca, E. L., Lamoreux, J. F., Michael, W. N. L., Lavilla, E. O., Leus, K., Lewison, R. L., Lichtenstein, G., Livingstone, S. R., Lukoschek, V., Mallon, D. P., Philip, J. K. M., McLvor, A., Moehlman, P. D. and Molur, S., 2010. The Impact of Conservation on the Status of the World's Vertebrates, *Science*, **330**(6010): 1503-1509. Available at: 10.1126/science.1194442

Howland, B. W. A., Stojanovic, D., Gordon, I. J., Fletcher, D., Snape, M., Stirnemann, I. A. and Lindenmayer, D. B., 2016. Habitat preference of the striped legless lizard: Implications of grazing by native herbivores and livestock for conservation of grassland biota, *Austral Ecology*, **41**(4): 461-470. Available at: 10.1111/aec.12337

Jackson, W., Argent, R., Bax, N., Clark, G., Coleman, S., Cresswell, I., Emmerson, K., Evans, K., Hibberd, M., Johnston, E., Keywood, M., Klekociuk, A., Mackay, R., Metcalfe, D., H, M., Rankin, A., Smith, D. and Wienecke, B., 2017. *Australia state of the environment 2016: overview*, Canberra.

Koh, N. S., Hahn, T. and Boonstra, W. J., 2019. How much of a market is involved in a biodiversity offset? A typology of biodiversity offset policies, *Journal of Environmental Management*, **232**: 679-691. Available at: 10.1016/j.jenvman.2018.11.080

Kutt, A. S., McKenzie, V. J., Wills, T. J., Retallick, R. W. R., Dalton, K., Kay, N. and Melero-Blanca, E., 2015. Spatial and temporal determinants of golden sun moth Synemon plana distribution, *Austral Ecology*, **40**(1): 100-107. Available at: 10.1111/aec.12181

Lindenmayer, D. and Fischer, J., 2006. *Habitat fragmentation and landscape change: an ecological and conservation synthesis*, Island Press, Washington.

Lindenmayer, D. B., Crane, M., Evans, M. C., Maron, M., Gibbons, P., Bekessy, S. and Blanchard, W., 2017. The anatomy of a failed offset, *Biological Conservation*, **210**: 286-292. Available at: 10.1016/j.biocon.2017.04.022

Maron, M., 2014. *Submission: Inquiry into Environmental Offsets*, Available at: https://www.aph.gov.au/Parliamentary_Business/Committees/Senate/Environment_and_Communications/Environmental_Offsets/Submissions

Maron, M., Brownlie, S., Bull, J. W., Evans, M. C., von Hase, A., Quétier, F., Watson, J. E. M. and Gordon, A., 2018. The many meanings of no net loss in environmental policy, *Nature Sustainability*, **1**(1): 19-27. Available at: 10.1038/s41893-017-0007-7

Maron, M., Bull, J. W., Evans, M. C. and Gordon, A., 2015. Locking in loss: Baselines of decline in Australian biodiversity offset policies, *Biological Conservation*, **192**: 504-512. Available at: https://doi.org/10.1016/j.biocon.2015.05.017

Maron, M., Dunn, P. K., McApline, C. A. and Apan, A., 2010. Can offsets really compensate for habitat removal? The case of the endangered red-tailedblack-cockatoo, *Journal of Applied Ecology*, **47**(2): 348-355. Available at: https://www.jstor.org/stable/40605827

Maron, M., Gordon, A., Mackey, B. G., Possingham, H. P. and Watson, J. E. M., 2016a. Interactions Between Biodiversity Offsets and Protected Area Commitments: Avoiding Perverse Outcomes, *Conservation Letters*, **9**(5): 384-389. Available at: doi:10.1111/conl.12222

Maron, M., Hobbs, R. J., Moilanen, A., Matthews, J. W., Christie, K., Gardner, T. A., Keith, D. A., Lindenmayer, D. B. and McAlpine, C. A., 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies, *Biological Conservation*, **155**: 141-148. Available at: 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., Gordon, A., Watson, J. E. M., Lentini, P. E., Gibbons, P., Possingham, H. P., Hobbs, R. J., Keith, D. A., Wintle, B. A. and Evans, M. C., 2016b. Taming a Wicked Problem: Resolving Controversies in Biodiversity Offsetting, *BioScience*, **66**(6): 489-498. Available at: 10.1093/biosci/biw038

Martin, N., Evans, M., Rice, J., Lodhia, S. and Gibbons, P., 2016. Using offsets to mitigate environmental impacts of major projects: A stakeholder analysis, *Journal of Environmental Management*, **179**: 58-65. Available at: 10.1016/j.jenvman.2016.04.054

Maseyk, F., Evans, M. and Maron, M., 2017. *Guidance for deriving 'Risk of Loss' estimates when evaluating biodiversity offset proposals under the EPBC Act*, Threatened Species

Recovery Hub,, Available at: https://tsrhub.worldsecuresystems.com/5.1 Guidance for deriving risk of loss Report_2017_LOW RES.pdf

Matthews, T. J., Cottee-Jones, H. E. and Whittaker, R. J., 2014. Habitat fragmentation and the species–area relationship: a focus on total species richness obscures the impact of habitat loss on habitat specialists, *Diversity and Distributions*, **20**(9/10): 1136-1146. Available at: 10.1111/ddi.12227

Maxwell, S., Fuller, R., Brooks, T. and Watson, J., 2016. The ravages of guns, nets and bulldozers, *Nature*, **536**(7615): 143-145.

May, J., Hobbs, R. J. and Valentine, L. E., 2017. Are offsets effective? An evaluation of recent environmental offsets in Western Australia, *Biological Conservation*, **206**: 249-257. Available at: 10.1016/j.biocon.2016.11.038

McCarthy, M. A., Parris, K. M., Van Der Ree, R., McDonnell, M. J., Burgman, M. A., Williams, N. S. G., McLean, N., Harper, M. J., Meyer, R., Hahs, A. and Coates, T., 2004. The habitat hectares approach to vegetation assessment: An evaluation and suggestions for improvement, *Ecological Management & Restoration*, **5**(1): 24-27. Available at: doi:10.1111/j.1442-8903.2004.00173.x

McLellan, R., Iyengar, L., Jeffries, B. and Oerlemans, N., 2014. *Living planet report 2014: species and spaces, people and places*, World Wide Fund for Nature,

Miller, K. L., Trezise, J. A., Kraus, S., Dripps, K., Evans, M. C., Gibbons, P., Possingham, H. P. and Maron, M., 2015. The development of the Australian environmental offsets policy: from theory to practice, *Environmental Conservation*, **42**(4): 306-314. Available at: 10.1017/S037689291400040X

Moilanen, A. and Kotiaho, J. S., 2018. *Planning biodiversity offsets : Twelve operationally important decisions*, Nordisk Ministerråd, Copenhagen.

Moreno-Mateos, D., Maris, V., Béchet, A. and Curran, M., 2015. The true loss caused by biodiversity offsets, *Biological Conservation*, **192**: 552-559. Available at: 10.1016/j.biocon.2015.08.016

Narain, D. and Maron, M., 2018. Cost shifting and other perverse incentives in biodiversity offsetting in India: Biodiversity Offsetting, *Conservation Biology*, **32**(4): 782-788. Available at: 10.1111/cobi.13100

Norton, D. A., 2009. Biodiversity Offsets: Two New Zealand Case Studies and an Assessment Framework, *Environmental Management*, **43**(4): 698-706. Available at: 10.1007/s00267-008-9192-5

Office of the Commissioner for Sustainability and the Environment, 2015. *ACT State of the Environment Report*, Canberra.

Parliament of Australia, 2014. *Inquiry into Environmental Offsets*, Available at: https://www.aph.gov.au/Parliamentary_Business/Committees/Senate/Environment_and_Communications/Environmental_Offsets

Peterson, I., Maron, M., Moillanen, A., Bekessy, S. and Gordon, A., 2018. A quantitative framework for evaluating the impact of biodiversity offset policies, *Biological Conservation*, **224**: 162-169. Available at: https://doi.org/10.1016/j.biocon.2018.05.005

Pickett, E. J., Stockwell, M. P., Bower, D. S., Garnham, J. I., Pollard, C. J., Clulow, J. and Mahony, M. J., 2013. Achieving no net loss in habitat offset of a threatened frog required high offset ratio and intensive monitoring, *Biological Conservation*, **157**: 156-162. Available at: 10.1016/j.biocon.2012.09.014

Pilgrim, J. D. and Bennun, L., 2014. Will Biodiversity Offsets Save or Sink Protected Areas?, *Conservation Letters*, **7**(5): 423-424. Available at: 10.1111/conl.12145

Pimm, S. L., Jenkins, C. N., Abell, R., Brooks, T. M., Gittleman, J. L., Joppa, L. N., Raven, P. H., Roberts, C. M. and Sexton, J. O., 2014. The biodiversity of species and their rates of extinction, distribution, and protection, *Science*, **344**(6187). Available at: 10.1126/science.1246752

Preston, F. W., 1962. The Canonical Distribution of Commonness and Rarity: Part I, *Ecology*, **43**(2): 185-215. Available at: 10.2307/1931976

Quigley, J. T. and Harper, D. J., 2006. Compliance with Canada’s Fisheries Act: A Field Audit of Habitat Compensation Projects, *Environmental Management*, **37**(3): 336-350. Available at: 10.1007/s00267-004-0262-z

Radford, J. Q., Bennett, A. F. and Cheers, G. J., 2005. Landscape-level thresholds of habitat cover for woodland-dependent birds, *Biological Conservation,* **124**(3): 317-337. Available at: 10.1016/j.biocon.2005.01.039

Richter, A., Osborne, W., Hnatiuk, S. and Rowell, A., 2013. Moths in fragments: insights into the biology and ecology of the Australian endangered golden sun moth Synemon plana (Lepidoptera: Castniidae) in natural temperate and exotic grassland remnants, *Journal of Insect Conservation,* **17**(6): 1093-1104. Available at: 10.1007/s10841-013-9589-1

SMEC Australia, 2015. *Golden Sun Moth population and habitat monitoring program,* Available at: https://www.environment.act.gov.au/__data/assets/pdf_file/0010/907138/Golden-Sun-Moth-Population-and-Habitat-Monitoring-2015-Report.pdf

SMEC Australia, 2017. *ACT Box - Gum Woodland Offset Monitoring*, Canberra ACT. Available at: https://www.environment.act.gov.au/__data/assets/pdf_file/0004/1096645/Box-Gum-Woodland-Monitoring-Report-2016.pdf (accessed January 2019).

Sonter, L. J., Tomsett, N., Wu, D. and Maron, M., 2017. Biodiversity offsetting in dynamic landscapes: Influence of regulatory context and counterfactual assumptions on achievement of no net loss, *Biological Conservation,* **206**: 314-319. Available at: https://doi.org/10.1016/j.biocon.2016.11.025

Spash, C. L. and Aslaksen, I., 2015. Re-establishing an ecological discourse in the policy debate over how to value ecosystems and biodiversity, *Journal of Environmental Management,* **159**: 245-253. Available at: https://doi.org/10.1016/j.jenvman.2015.04.049

Sudol, M. F. and Ambrose, R. F., 2002. The US Clean Water Act and Habitat Replacement: Evaluation of Mitigation Sites in Orange County, California, USA, *Environmental Management,* **30**(5): 0727-0734. Available at: 10.1007/s00267-002-2787-3

Sullivan, R., 2005. *Rethinking voluntary approaches in environmental policy*, Edward Elgar, North Hampton, MA;Cheltenham, UK;.

Thorn, S., Hobbs, R. J. and Valentine, L. E., 2018. Effectiveness of biodiversity offsets: An assessment of a controversial offset in Perth, Western Australia, *Biological Conservation,* **228**: 291-300. Available at: 10.1016/j.biocon.2018.10.021

Energy, D. o. E. a., *Commonwealth Listing Advice on White Box-Yellow Box-Blakely's Red Gum Grassy Woodland and Derived Native Grassland,*

Tischew, S., Baasch, A., Conrad, M. K. and Kirmer, A., 2010. Evaluating Restoration Success of Frequently Implemented Compensation Measures: Results and Demands for Control Procedures, *Restoration Ecology,* **18**(4): 467-480. Available at: 10.1111/j.1526-100X.2008.00462.x

Tulloch, A. I. T., Barnes, M. D., Ringma, J., Fuller, R. A., Watson, J. E. M. and Mori, A., 2016. Understanding the importance of small patches of habitat for conservation, *Journal of Applied Ecology,* **53**(2): 418-429. Available at: 10.1111/1365-2664.12547

Umwelt (Australia) Pty Limited, 2013. Gungahlin Strategic Assessment - Biodiversity Plan. Available at: http://www.environment.gov.au/system/files/pages/b58dc6ac-a4f2-4a9e-9dad-4f0752e0f050/files/gungahlin-biodiversity-plan_0.pdf

United Nations, D. o. E. a. S. A., Population Division,, 2017. *World Population Prospects: The 2017 Revision, Key findings and advance tables,*

United Nations, D. o. E. a. S. A., Population Division,, 2018. World Urbanization Prospects: The 2018 Revision, Key facts.

Watson, J. E. M., Dudley, N., Segan, D. B. and Hockings, M., 2014. The performance and potential of protected areas, *Nature,* **515**(7525): 67-73. Available at: 10.1038/nature13947

Wilkins, S., Keith, D. A. and Adam, P., 2003. Measuring Success: Evaluating the Restoration of a Grassy Eucalypt Woodland on the Cumberland Plain, Sydney, Australia, *Restoration Ecology,* **11**(4): 489-503. Available at: 10.1046/j.1526-100X.2003.rec0244.x

Yencken, D. and Wilkinson, D., 2000. *Resetting the compass: Australia's journey towards sustainability,* CSIRO Publishing, Collingwood, Vic.

Appendix 1: Included projects

List of developments (and associated offset sites) included in this study as per the ACT Government's offset register with given unique identifiers.

Development	Related offsets
Gungahlin District development (strategic assessment)	1A. Horse Park North Conservation Area 1B. Jacka Conservation Area 1 1C. Jacka Conservation Area 2 1D. Kenny Broadacre Conservation Area 1E. Kenny Conservation Area 1F. Kinlyside Conservation Area 1G. Taylor Conservation Area 1H. Throsby East Conservation Area 1I. Throsby North Conservation Area
Molonglo Valley development (strategic assessment)	2A. Glenloch Woodland (Patch GG & N) 2B. Kama Nature Reserve 2C. Molonglo River Corridor
Campbell Section 5	3A. Yarralumla Equestrian Park Offset Area
Macgregor West 2 Estate	4A. Macgregor West 2 Estate Offset Area
Ngunnawal Residential Estate Stage 2C	5A Bonner 4 East Offset Area
Mugga Resource Management Centre expansion	6A. Isaacs Ridge
Block 9 Section 64 Watson and Negus Cres extension	7A. Watson Woodlands
EPIC Block 799 Cabin and Camping development	8A. Gungaderra Grasslands Nature Reserve Extension 8B. Mulanggari Grasslands Nature Reserve Extension
University of Canberra Public Hospital	9A. Pinnacle Nature Reserve Offset Site
Lawson south residential development, Belconnen	10A. Jarramlee Offset Site

Appendix 2: Excluded projects

List of offsets (and associated development sites) that are excluded from this study yet listed on the ACT Government's offset register

Offset/s	Associated Development	Approval date	Reason for exclusion
Kings Highway Offset	Deviation of Kings Highway, Kowen (2010/5501)	25/08/2010	1. Impacts largely indirect and not easily quantified (2.6 ha of direct removal, 44 ha of fragmentation) 2. No documents available on offsets register
Williamsdale site 2 Offset Area	Electricity substation and access road, Williamsdale (2009/4805)	21/05/2009	1. No referral documents available on EPBC website 2. No documents available on offsets register
Williamsdale site 3 Offset Area	132Kv Sub-transmission line Williamsdale to Theodore (2008/4621)	07/08/2009	1. No referral documents available on EPBC website 2. No documents available on offsets register
Williamsdale site 1 Offset Area Williamsdale site 4 Offset Area	Murrumbidgee to Googong water transfer and associated infrastructure (2009/5124)	29/10/2010	1. No referral documents available on EPBC website 2. Offset management plan available on offset register website however details of the impact area are not included. In order to remain consistent with exclusion of the other Williamsdale offset sites, and due to the lack of detail regarding development impacts, this was also excluded.
Kama Nature Reserve Revegetation Area	Clarrie Hermes Drive Extension, West Gungahlin (2009/5156)	27/10/2010	1. Impacts largely indirect and not easily quantified (1.77 ha of direct removal, 30 ha of isolation) 2. Offset is indirect (revegetation in existing nature reserve) 3. No referral documents available on EPBC notices website

Appendix 3: Vegetation classification for analysis

Classification of vegetation types for connectivity analysis

Native

Apple Box – Broad-leaved Peppermint tall shrub-grass open forest

Aquatic fringing vegetation

Black Cypress Pine – Brittle Gum tall dry open forest on hills

Blakely's Red Gum – Yellow Box tall grassy woodland

Blakely's Red Gum – Yellow Box tall grassy woodland

Derived native shrubland

Drooping She-oak low woodland to open forest on shallow infertile hillslopes in the Australian Capital Territory and surrounds

Environmental planting native

Mealy Bundy – Red Stringybark grass-forb mid-high open forest

Native grassland

Natural Temperate Grassland

Red Box tall grass-shrub woodlands primarily on hillslopes and footslopes in the Australian Capital Territory

Red Stringybark – Scribbly Gum – Red-anthered Wallaby Grass tall grass-shrub dry sclerophyll open forest on loamy ridges

River She-oak riparian forest on sand-gravel alluvial soils along major watercourses

Snow Gum grassy mid-high woodland

Yellow Box ± Apple Box tall grassy woodland

Excluded

Amenity planting exotic

Amenity planting native*

Arboriculture

Exotic forest

Exotic grassland

Exotic shrubland

Exotic woodland

Plantation exotic

Power easement

Urban and developed areas

Urban Open Space

Water

* Amenity planting native was excluded from the 'native' category on the basis that: (1) the species planted are not necessarily those that are native to the ACT and (2) the structure of amenity planting favours the practicality of the built space, as such, the habitat value of these areas is usually poor (i.e. due to mowing and other disturbances).

Appendix 4: Key assumptions in no net loss analysis

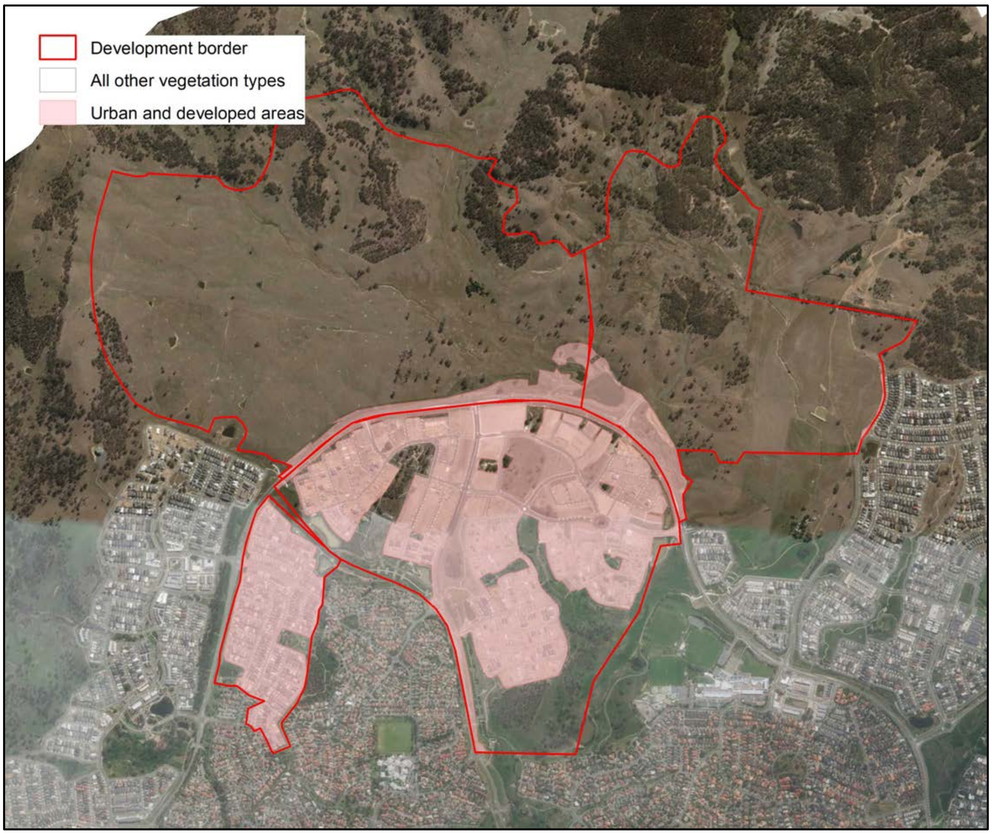
Key assumptions/biases in the no net loss analysis, justification and considerations regarding these and the expected 'impact' on the results

Assumption	Justification/considerations	Impact
Vegetation lost due to development and vegetation offset occur simultaneously within year of EPBC Act approval	Difficulty determining when vegetation was lost, and when offsets were secured.	Time until no net loss may be a few years longer than that estimated, depending on how long after development offsetting occurred (including restoration measures).
Area is assumed to represent the overall losses and gains in native vegetation associated with development and offsetting (i.e. sites do not vary in quality/biodiversity value).	Comparable quality data through time was not available. This particular assumption was also made by Gibbons et al. (2018) in their analysis.	Vegetation quality is not represented in the analysis. Meaning that, analysis is simplified, and may not represent true gains expected from restoration (if a significantly higher quality is achieved).
The scenarios simulated have various proportions of overall gains associated with offsets (averted loss versus restoration) and <u>do not</u> overlap	A recent monitoring report suggests that it is likely that the offsets reviewed here included a mix of averted loss gains, and restoration gains. This is because the 'conservation value' of the offset sites assessed by this report varied greatly, both within the sites themselves, and in comparison to each other (ranging from 'low to exceptional' (SMEC Australia, 2017). However, due to the lack of specific reporting/data available on offset calculations during assessment and approvals process, I was unable to conclude exact proportions, and so, the accuracy of these is unknown. In addition, there is the potential that gains associated with averted loss and restoration are	No net loss analysis could under or overestimate years before no net loss is achieved depending on how offsets procured gains in reality (proportion of averted loss and restorative measures).

	overlapping in nature. That is, the securing of an offset area could be considered to avert losses while also include the calculation of gains associated with restoration actions applied to the site.	
Restoration does not accrue over time	To estimate the timeframe over which restoration gains were expected to accrue was not possible using the data available. Assuming a timeframe was considered (10-20 years) however, this would have added a considerable amount of uncertainty to the analysis. By not assuming the period over which restoration gains would be realised, I was then able to discuss how the no net loss analysis results would change depending on this time-frame.	Results are likely to under-estimate the years before no net loss is achieve. However, this difference is equal to the amount of years before restoration gains are realised.
The risk of loss reflects that of the overall box gum woodland rate of decline in the ACT	It was not feasible to determine the risk of loss for each individual offset, and further, advice for calculating this percentage is often vague (for example see EPBC offset assessments guide) (Australian Government, 2012b). This means that, the actual risk of loss (or site counterfactual scenario) for each offset could have been calculated in varying ways. As such, the most unbiased approach available was to use the overall rate of decline of the vegetation type that represents the majority of vegetation offset. This is box gum woodland, which accounts for 899.4 ha out of 1329.4 ha of vegetation offset in the ACT.	Using this method means that the results may over-estimate the years before no net loss if a higher risk of loss (counterfactual scenario) were applied during assessment.
'Restoration success' is a fixed and measurable variable	There was not scope within this project to assess each offset against a set criteria to determine the overall success of restoration. While there have been a few reports in recent years that quantify restoration effectiveness based on generalised criteria (e.g. May *et al.*, 2017), it was not possible to determine restoration success from available documentation (and considering the relatively short time period these offsets have been under management).	The application of 'restoration success' in the analysis is simple, where restoration success (%) translates to the amount of area gained through offsetting (rather than 'gains' associated with the improvement of a site's condition.

Appendix 5: Example of corrections made to development sites in Arcmap

Example of development site where vegetation map reflects development impacts (shown in pink – ‘urban developed areas’), and where developed is planned but yet to occur (area within ‘development border’ which is not shaded pink).



Appendix 6: Impacts and offset areas by project

Summary of impact and offset areas associated with each development (as per assessment/approval documentation)

Development	Impact	Offset
Block 9 Section 64 Watson and Negus Cres extension	4.00	16.00
Campbell Section 5	3.00	7.60
EPIC Block 799 Cabin and Camping development	14.65	44.40
Gungahlin District Development	326.00	863.00
Lawson south residential development, Belconnen	38.35	53.00
Macgregor West 2 Estate	12.00	37.00
Molonglo Valley	137.00	234.00
Mugga Resource Management Centre expansion	9.80	36.90
Ngunnawal Residential Estate Stage 2C	14.80	21.00
University of Canberra Public Hospital	7.60	15.50
Grand Total	**567.20**	**1328.40**

Appendix 7: Vegetation types within development sites and offset sites

Vegetation types as a % of the total (vegetated) area within development sites and offset sites and total ha (presented in Figure 7)

Vegetation type	Development sites (ha)	Offset sites (ha)
Native (total)	**1559.07**	**1494.83**
Derived native shrubland	2.33	21.30
Mealy Bundy – Red Stringybark grass-forb mid-high open forest	15.97	68.78
Native grassland	1193.21	688.72
Natural temperate grassland	30.83	110.21
Yellow Box - Apple Box tall grassy woodland	29.13	92.86
Blakely’s Red Gum – Yellow Box grassy woodland	257.42	390.80
Red Stringybark – Scribbly Gum – Red-anthered Wallaby Grass tall grass-shrub dry sclerophyll open forest on loamy ridges	30.18	25.68
River She-oak riparian forest on sand-gravel alluvial soils along major watercourse	0	96.48
Exotic (total)	**394.68**	**122.84**
Exotic forest	107.71	12.63
Exotic grassland	59.71	44.55
Exotic shrub land	25.98	9.06
Exotic woodland	6.02	31.10
Plantation exotic	195.26	25.51
Total	**1953.75**	**1617.68**

Appendix 8: Additional land (ha) allocated Nature Reserves and offsets before and after the introduction of offsetting in the ACT

Summarised totals (ha) that correspond with Figure 11, which demonstrate additional land allocated before (2004-2009) and after (2010-2014) the introduction of offsetting.

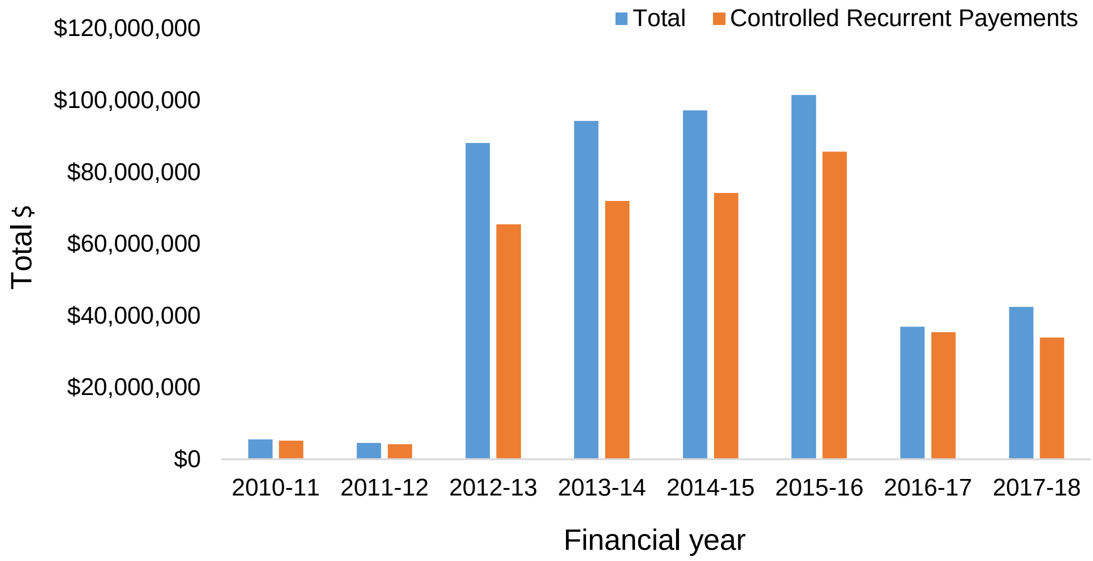
Allocation type	**2004-2009** **(5 years before offsetting)**	**2010-2014** **(5 years after offsetting)**
Nature Reserves (excluding area reserved for offsets)	1495 ha	445.68 ha
Offsets formally reserved	0 ha	308 ha
Offsets	0 ha	1328.40 ha

Appendix 9: CAPAD data summary

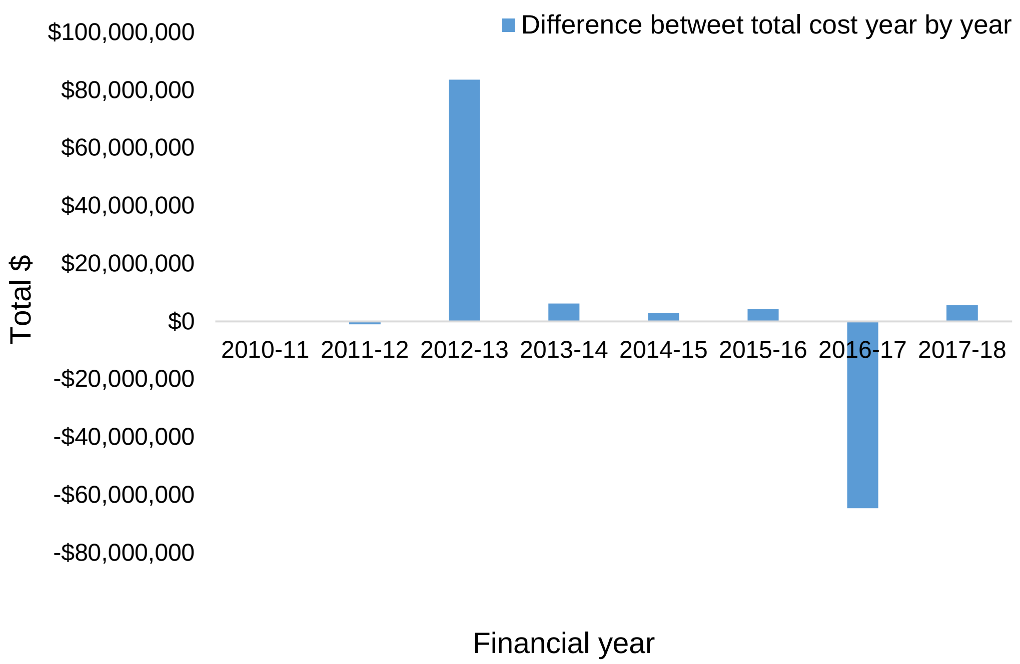
Additional area reserved each year in the ACT based on CAPAD data (by gazettal date, not date reported) and not including offsets.

Year	Nature Reserve (ha)
1993	30316.3
1994	791.09
1995	554
1996	0
1997	105
1998	0
1999	0
2000	0
2001	0
2002	0
2003	0
2004	0
2005	0
2006	781
2007	0
2008	714
2009	0
2010	0
2011	0
2012	112.68
2013	19
2014	314
Grand Total	**33707.07**

Appendix 10: Financial analysis

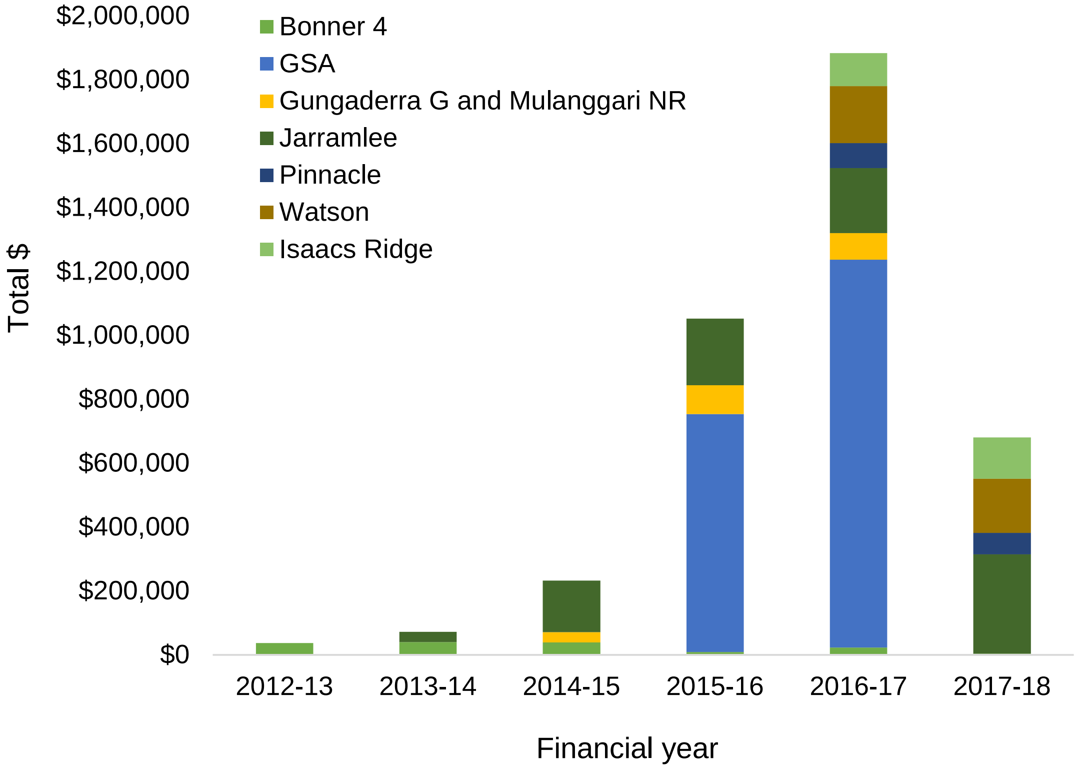


Estimated outcome per year for Conservation and Land Management in the ACT, based on budget statements available at: (https://apps.treasury.act.gov.au/budget/budget-2016-2017/budget-statements).



Difference in total budget per year for Conservation and Land Management in the ACT, based on budget statements available at:

(https://apps.treasury.act.gov.au/budget/budget-2016-2017/budget-statements).



Total allocation to offsets (by development project) per year based on compliance reports available via the ACT offsets register.

Appendix 11: Guidance on calculating site counterfactuals

Box 1 | The problem with including type 1 impacts in counterfactuals

Type 1 impacts are those that trigger an impact-specific NNL policy; type 2 impacts are those that do not. In a hypothetical landscape, a threatened plant population (see photo) is declining due to two factors: impacts from mining and livestock grazing. A NNL policy that aims to counterbalance impacts on threatened species applies to all new impacts from mining, but not to the ongoing impacts of grazing.

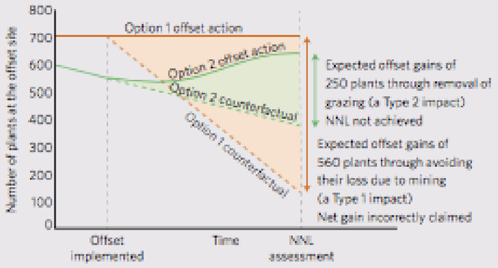
Company X submits plans for a new mine that will impact 500 of the remaining threatened plants. It has two options to offset this impact (see figure). Option 1 involves protecting another part of the mining lease, which supports 700 individuals of the same plant, but might otherwise be mined in the future, resulting in the plants being lost. Option 2 is to purchase an adjoining property that has 600 of the threatened plants, but is subject to livestock grazing. Company X would remove the grazing in the hope that this will increase the plant population.

Company X proposes that option 1 would achieve a net gain outcome under the NNL policy. Their calculation relies on a counterfactual scenario for the site: how many plants there would be if the site did not become an offset. They state that if they were not to protect this part of their lease through an offset, there is a high chance — estimated at 80% — that the site would be lost to mining (a type 1 impact), resulting in loss of all of the threatened plants. The expected loss of plants without the offset is therefore 0.8×700 plants. By protecting the site from mining, however, all 700 plants would remain; company X therefore concludes that the offset benefit of avoiding the loss of 560 plants more than counterbalances the original impact (loss of 500 plants) and achieves NNL.

It is not valid for company X to claim the benefit from the avoided loss of the offset site to mining (a type 1 impact) because, according to the policy, any future mining at the site would also have been subject to a NNL requirement, and thus its own offset. The loss of the site would have to be counterbalanced elsewhere, with a gain of 700 plants required. Thus, the actual benefit of option 1 is zero.

Option 2, however, is a different story. The continuation of livestock grazing (a type 2 impact) will cause the loss of 200 of the threatened plants, and its removal is expected to increase the population to 650. So, the benefit of Option 2 is avoidance of the loss of 200 plants, plus the increase of 50 plants — a total benefit of 250 plants that would not otherwise exist. Option 2 provides only half of the benefit required for a NNL outcome, meaning that company X would need to implement additional offsets — but it is a much more beneficial offset than option 1, which incorrectly included the avoidance of type 1 impacts in their calculation of benefit.

An hypothetical plant species threatened by both Type 1 and Type 2 impacts



Example of the problem with including ‘type 1 impacts’ (i.e. impacts covered by the offset policy), where net gain is incorrectly gained and this difference is demonstrated in the graph (red), as opposed to the correct calculation (green) which yields less gains overall.